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13. ABSTRACT *(Maximum 200 words)* Contaminated groundwater, which contained multiple heavy metals and chlorinated aliphatic hydrocarbons, from the surficial aquifer (well CC-33B) at Beach Point located in the Canal Creek Area of the U.S. Army Aberdeen Proving Ground-Edgewood Area, Aberdeen, Maryland, was evaluated for toxicity and environmental hazard. Toxicity was detected at various groundwater concentrations by 7 of 9 biomonitoring systems. When estimated maximum acceptable toxicant concentrations (MATC) were established, the data for algae, invertebrates and fish suggested that the groundwater would not be harmful at a concentration of 10% groundwater by volume. Likewise, no genotoxicity (Ames and SEC assays), developmental toxicity (FETAX), or chronic histopathology (9-month fish test) occurred at 10% groundwater by volume.

The groundwater was considered to be a potentially excessive hazardous material to the benthic biota of the Bush River when a number of conservative assumptions (contaminant distribution and discharge rate of the aquifer) were used in the hazard assessment. However, the potential water quality impacts were judged to be minimal if a mixing zone were to be granted by the State of Maryland which allows for local exceedences of water quality standards. (Cont. on reverse)

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Near-field (ULINE model) and far-field (dye-tracer model) screening level dilution models were run to estimate the dilution of the groundwater discharge plume in the Bush River. The model analyses showed that a near-field dilution level of approximately 42:1 for the application of Maryland's acute aquatic life criteria and a near-field dilution level of 168:1 for the application of chronic criteria would occur. When the dilution factors were applied to groundwater quality at Beach Point, none of the heavy metals or chlorinated aliphatic compounds exceeded Maryland's current acute or chronic aquatic life criteria.

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P.T. Burton
Principal Investigator's Signature

11-18-94
Date

EXECUTIVE SUMMARY

The primary objective of the study was to evaluate the use of several biomonitoring systems for assessing contaminated groundwater. Contaminated groundwater, which contained multiple heavy metals and chlorinated aliphatic hydrocarbons, was studied. The contaminated groundwater was withdrawn from the lower depths of the surficial aquifer at Beach Point which is located in the Canal Creek Area of the U.S. Army Aberdeen Proving Ground-Edgewood Area, Aberdeen, MD. Groundwater was withdrawn from well CC-33B, which is the most the highly contaminated well at Beach Point. A hazard assessment of the groundwater release into the Bush River was performed using the biomonitoring results.

Biomonitoring Evaluation

Several EPA priority pollutant heavy metals were found in the groundwater at Beach Point. The concentrations of cadmium, copper, nickel, and zinc exceeded one or more of EPA's numerical water quality criteria for the specific metal. A number of chlorinated aliphatic compounds were also found in the groundwater, several of which are EPA priority pollutants. The chlorinated aliphatic organics that occur at the lower depths of the aquifer may be present as a residual denser-than-water nonaqueous phase liquid (DNAPL). Of the organics present in the groundwater, none had octanol water partition coefficients (log k_{ow} or log P) >3; thus, bioaccumulation was not a potential toxicological problem.

An array of nine biomonitoring systems integrated into a tiered hazard framework was evaluated during a 9-month study. The biomonitoring systems included a number of endpoints. The pH of the groundwater from well CC-33B was 4 (± 0.1); thus, many of the assays were conducted at both pH 4 and pH 7. The toxicity at pH 7 was studied so that the data could be used in the hazard assessment of the groundwater as it entered the Bush River which has pH values close to the neutral range.

Toxicity was detected at various groundwater concentrations by 7 of 9 biomonitoring systems. The Ames assay for mutagenicity was negative in all cases (1, 10, and 100% groundwater by volume). Japanese medaka (Oryzias latipes) growth was not affected by 9 months of exposure to 1 and 10% groundwater by volume in a chronic histopathology assay. No significant lesions were found in the Japanese medaka exposed to groundwater concentrations up to 10% groundwater by volume (highest concentration studied). A positive response was found for 100% groundwater in a sister chromatid exchange (SCE) assay for DNA damage when the groundwater sample was concentrated ≈50,000X. The SCE assay was negative for unconcentrated 100% groundwater. The positive SCE response in the 50,000X concentrated sample was

judged not to be important to aquatic organisms in the receiving stream.

The lowest concentration of groundwater that caused no observable adverse effect (NOEC) at pH 4, in the test systems in which the NOEC value could be determined, was 10% groundwater by volume. A NOEC of 10% groundwater by volume occurred in 3 out of 4 tests in both a 7-d cladoceran (Ceriodaphnia dubia) and a 96-h frog (Xenopus laevis) embryo teratogenesis assay - Xenopus (FETAX); it occurred once in a 7-d fathead minnow (Pimephales promelas) test. The NOEC concentration was higher at pH 7 in both the fathead minnow and FETAX assays. The 10% groundwater by volume NOEC for the cladoceran at pH 4, however, did not change when the organism was exposed to buffered groundwater at pH 7.

When estimated maximum acceptable toxicant concentrations (MATC) were established, the data for alga (Selenastrum capricornutum), invertebrate (C. dubia), and fish (P. promelas) used in the biomonitoring study suggested that the groundwater would not be harmful at a concentration of 10% groundwater by volume. Likewise, no genotoxicity, developmental toxicity, or chronic histopathology occurred at 10% groundwater by volume. Thus, the biomonitoring data suggested that chronic toxicity may not occur in the Bush River at a dilution of 10:1. Chronic toxicity was predicted to occur in the Bush River if the groundwater entered the receiving stream at the concentrations which occur in well CC-33B.

Hazard Assessment Evaluation

A number of worst-case assumptions were made for the Beach Point hazard assessment when sufficient data were not available. The flowing assumptions were made for the calculation of the estimated discharge rate of the surficial aquifer into the Bush River. The surficial aquifer was considered to be homogeneous, characterized by isotropic flow conditions. The average saturated thickness of the aquifer was assumed to be 18.8 m (61.8 ft). The aquifer was assumed to discharge the entire length of the Beach Point Peninsula fronting the Bush River; a more realistic estimate of length of discharge from Beach Point is probably one third to one half, rather than the entire length.

With regard to the contaminant concentrations in the groundwater, the assumption was made for heavy metals that no retardation occurred via adsorption onto solid surfaces or trapping by clays through ion exchange. It was also assumed that no precipitation of the metals occurred when the pH shifted from 4 to neutrality when the groundwater entered the receiving stream. The assumption was made for the DNAPLs that no abiotic (chemical) or biotic (microbial) transformations occurred.

It was also assumed that the heavy metals and residual DNAPLs were homogeneously mixed throughout the complete aquifer at the highest concentration (not the average concentration) measured during the course of the study. It was assumed that the maximum concentrations of materials would not increase above those currently present in well CC-33B since the original sources of the contaminants were no longer present. The assumption was made that the highest concentrations of the heavy metals and chlorinated aliphatics in the aquifer all moved through the sediments into the Bush River without any biotransformations or other physical/chemical transformations occurring in the sediment or water column of the Bush River.

When the above assumptions were used in the hazard assessment, the groundwater may be considered to be a potentially hazardous material to the benthic biota of the Bush River. The hazard to water column aquatic biota would rapidly dissipate as the groundwater materials mix in the receiving stream. Because the potential water quality impacts were judged to be minimal, a mixing zone approach by the State of Maryland which allows for local exceedences of water quality standards may be pursued. The State of Maryland Code (COMAR) allows for a mixing zone on a case-by-case basis. Pollutant concentrations within a mixing zone may exceed the specified water quality standards within a localized vicinity of an outfall.

Near-field (ULINE model) and far-field (dye-tracer model with input data originally obtained for the Bush River) EPA-approved screening level dilution models were run to estimate the dilution of the groundwater discharge plume in the Bush River. The concentrations of the groundwater hazardous substances were estimated for the near-field and far-field in the Bush River using the mixing zone requirements of the State of Maryland.

The model analyses showed that a total near-field dilution (near-field dilution corrected for the influence of far-field dilution) of approximately 42:1 for the application of the State's acute aquatic life criteria and a near-field dilution level of 168:1 for the application of chronic criteria would occur. Thus, contaminants introduced via Beach Point groundwater into Bush River receiving waters at a concentration of 42 μg/L would be diluted locally to a concentration of approximately 1 μg/L or less. When the dilution factors were applied to groundwater quality at Beach Point, none of the heavy metals or chlorinated aliphatic compounds exceeded Maryland's current acute or chronic aquatic life criteria. The dilution study showed that detectable concentrations of some of the chlorinated organics would occur in the receiving stream when the conservative assumptions concerning the groundwater contaminants were used in the model.

Although an argument can be made for a mixing zone, it is highly unlikely that the contaminant loads assumed in the near-field dilution model will ever occur in the Bush River. A refinement of the conservative assumptions used in the hazard assessment would reduce the uncertainty regarding the volume of the discharge as well as the concentration of contaminants that may enter the receiving stream.

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SECTION 1

INTRODUCTION

The primary goal of ecotoxicological testing is to predict the effects of chemicals (single elements, compounds, and mixtures) and other stressors (e.g., heated wastewater, suspended solids, etc.) on the long-term health of individual organisms, populations, communities, and ecosystems. There is no perfect short-term (acute) or long-term (chronic) toxicity test which allows one to predict with certainty the effects of a single toxicant (not to mention a complex mixture of contaminants) on ecosystems. Although perfect prediction is not attainable (Barnhouse et al., 1986), a well selected array or battery of biomonitoring assays integrated into a tiered hazard assessment approach can be used to maximize predictability of adverse pollutant effects to aquatic ecosystems (for ex., see Dutka and Kwan, 1988; National Research Council, 1981; Schaeffer and Janardam, 1987).

The selection of an appropriate tier of hazardous assessment biomonitoring tests is based on the problem to be solved (Cairns, 1990; Dickson et al., 1979; Mackay et al., 1989). Single species assays are the most widely used hazard assessment systems (for ex., see Cairns, 1986; Herricks and Schaffer, 1985; Rand and Petrocelli, 1985). Hazard assessment schemes may also incorporate multispecies microcosm tests and ecosystem level simulations in their protocols. Single species toxicity tests, multispecies microcosm tests, and ecosystem level simulations all have advantages and limitations (Neuhold, 1986). The U.S. Army Biomedical Research and Development Laboratory's (USABRDL) Research Methods Branch uses a hazard assessment biomonitoring testing tier which is composed primarily of an array of single species assays.

The University of Maryland in collaboration with USABRDL is evaluating the use of several biomonitoring systems for assessment of potentially contaminated groundwater and sediment-associated contaminants at U.S. Army installations. The current report summarizes an evaluation of contaminated groundwater which contained multiple heavy metals and chlorinated aliphatic hydrocarbons. The contaminated groundwater was withdrawn from the surficial aquifer at Beach Point which is located in the Canal Creek Area of the U.S. Army Aberdeen Proving Ground-Edgewood Area, Aberdeen, MD. The definitive experimental phase of the study was conducted over a 9-month period from February 25, 1993 to December 6, 1993. In addition to the primary goal of the study to evaluate the performance of the biomonitoring systems, the biomonitoring results were also used in an hazard assessment of the groundwater release in the Bush River.

SECTION 2

OBJECTIVES OF STUDY

The primary objectives of the study were to:

1) Evaluate the acute toxicity of the groundwater using the 5- and 15-min Microtox® procedure (Photobacterium phosphoreum bioluminescent activity) and the 24-h LC50 Rotifer Toxkit™ (Brachionus rubens) screening test. In addition, the following acute bioassays were also performed: algal (Selenastrum capricornutum) 96-h EC50 growth test, cladoceran (Ceriodaphnia dubia) 48-h LC50, fathead minnow (Pimephales promelas) 96-h LC50, and Japanese medaka (Oryzias latipes) 96-h LC50.

2) Evaluate chronic toxicity using the 96-h EC50 algal (S. capricornutum) growth test, 7-d cladoceran (C. dubia) survival and reproduction test, and 7-d fathead minnow (P. promelas) survival and growth test.

3) Determine the genotoxicity potential of unconcentrated and concentrated samples of the groundwater using the Ames assay and Sister Chromatid Exchange assay.

4) Determine the developmental toxicity potential of the groundwater using the frog (X. laevis) embryo teratogenesis assay - Xenopus (FETAX).

5) Determine the chronic histopathological potential of the groundwater using a 9-month Japanese medaka (O. latipes) growth and chronic histopathology test.

6) Quantify the major chemicals present in the groundwater and monitor the general water quality of the groundwater.

A secondary objective of the study was to use the biomonitoring data obtained in the biomonitoring system evaluations to assess the potential hazard of the groundwater as it moved into an aquatic ecosystem (Bush River).

SECTION 3

BEACH POINT SITE DESCRIPTION

3.1 Physiographic Setting

Beach Point is a peninsula located in the Canal Creek Area of the U.S. Army Aberdeen Proving Ground-Edgewood Area (APG-EA), Aberdeen, MD. The peninsula is bounded by the Bush River on the southeast and Kings Creek on the northwest. The long axis of the peninsula, which is oriented approximately 45° east of north, is ≈448 m (≈1,470 ft) long x ≈107 m (≈350 ft) wide. The maximum altitude of the land surface is ≈4.6 m (≈15 ft) above mean sea level (Nemath, 1989); the average elevation is ≈3 m (≈10 ft) above mean sea level (McGinnis et al., 1994).

3.2 Historical Use

Beach Point has been used for a number of activities which may have contributed to the contamination of the soils and groundwater as well as Kings Creek and Bush River (Nemath, 1989). The major operations include 1) mobile and fixed-based clothing impregnating plant activities during World War II; 2) liquid rocket fuel testing from the early 1960s through the 1970s; and 3) pyrotechnic testing with grenades and pots filled with obscurant smoke from the post-World War II period to about 1970.

As discussed in Nemath (1989), the clothing impregnating process involved the use of several solvents which included 1,1,2,2-tetrachlorethane, N,N' dichloro-bis(2,4,6-trichlorophenyl)urea, chlorobenzene, and other solvents. Losses of the hazardous materials may have occurred through volatilization, spillage, or leakage, as well as direct discharge to Kings Creek and/or the Bush River. Historical photographs indicate that the materials may have also been discharged to small pits located near the clothing impregnating plants. Various fuels, oxidizers, and fire suppressants were used in the liquid rocket fuel testing programs. Wastewater from the tests was either discharged directly to the Bush River and/or Kings Creek, or may have been allowed to run off onto the ground surface. The primary materials used in the pyrotechnic testing were white obscurant smokes and fog oil. Materials associated with pyrotechnic testing often include aluminum, magnesium, zinc, lead, and titanium, as well as petroleum compounds, hexachloroethane, and other organic compounds (JEG, 1993).

3.3 Groundwater Contamination

Because of possible groundwater contamination from historical activities at Beach Point, USGS began placing observation wells at Beach Point in 1986 as part of a larger

study of the Canal Creek area (Lorah and Vroblesky, 1989). Five wells, located in three clusters, were originally drilled in the surficial aquifer (see Section 3.4) at Beach Point during 1986 in the proximity of the former clothing impregnating plants. A sixth well (CC-33B.1) was drilled in 1988. The three clusters contain the following wells. Cluster 32 consists of wells CC-32A and CC-32B which have screened depths below land surface of 3.2-4.7 and 6.4-7.9 m (10.5-15.5 and 21-26 ft), respectively. Wells 33A, 33B, and 33B.1 in cluster 33 have screened depths of 3.4-4.9, 18.9-20.4, and 12.5-14.0 m (11-16, 62-67, and 41-46 ft), respectively. Well 34A, with a screen interval of 4.3-5.8 m (14-19 ft), is the only well in cluster 34. Of the six monitoring wells at Beach Point in the surficial aquifer, four are shallow with screen depths that range from 3.2-7.9 m (10.5-26 ft), one well (CC-33B.1) is intermediate with a screen depth of 12.5-14.0 m (41-46 ft), and one well (CC-33) is deep with a screen depth of 18.9-20.4 m (62-67 ft).

Chemical monitoring (inorganic and organic constituents) by USGS during 1988 confirmed that hazardous chemicals from prior activities were present in the groundwater (Lorah and Vroblesky, 1989 and Lorah and Clark, 1992). Several inorganic constituents exceeded Federal drinking water standards; the highest concentrations were found in the deep well (CC-33B). Several heavy metals were also found in the wells. The highest concentrations were generally found in wells CC-33B and CC-33B.1; the concentrations of heavy metals in wells CC-33B and CC-33B.1. were similar. A number of volatile chlorinated aliphatic compounds were found in the groundwater at Beach Point. With the exception of chloroform in well CC-32A and methylene chloride in wells CC-33A and CC-33B.1, the highest concentrations of the volatile organics were found in the deep well CC-33B. Wells CC-33A and CC-33B only were studied at Beach Point during 1989. The concentrations of heavy metals and volatile organics were all higher in the deep well (CC-33B) than in the shallow well (CC-33A).

The chlorinated aliphatic organics that occur at the lower depths of the aquifer may be present as a residual denser-than-water nonaqueous phase liquid (DNAPL) (Cohen and Mercer, 1993; Knox et al., 1993). McGinnis et al. (1994) recently found that the surficial aquifer at Beach Point is a relatively low-resistivity lens pierced by high resistivity "plumes" that may indicate the presence of DNAPLs. Additional chemical data are needed from the high resistivity plumes in areas other than well CC-33B to confirm the presence of DNAPLs.

3.4 Geology

The geology of the Canal Creek area has been described by USGS (Oliveros and Vroblesky, 1989). JEG (1993) has recently summarized the geology of the Beach Point study site. Briefly,

the Aberdeen Proving Ground-Edgewood Area is underlain by coastal plain sediments consisting of unconsolidated clay, silt, and sand layers with occasional gravel lenses. At Beach Point the surficial sediments consist of a clayey soil to a depth of ≈1.2 m (4 ft), underlain by ≈18.2 m (≈60 ft) of fine- to medium-grained, well-sorted sand, interfingered with thin lenses of clay and silty sand and sometimes containing coarse sand and gravel layers (Oliveros and Vroblesky, 1989). The surficial unit, which appears to be part of the Potomac Group, is underlain by a clay layer called the upper confining unit. According to JEG (1993), the upper confining unit, which has a thickness of ≈27 m (88 ft), does not appear to have been penetrated at Beach Point. Thus, it is unlikely that the Canal Creek Aquifer below the upper confining unit has been contaminated by activities at Beach Point. The Beach Point surficial sediments and clay upper confining unit appear to dip gently (≈15 m/mile; 50 ft/mile) to the southeast.

3.5 Groundwater Hydrology

Groundwater at Beach Point is encountered at shallow depths from less than 4 m (13 ft) to about 4.9 m (16 ft) below ground surface in most areas under unconfined conditions (JEG, 1993). The surficial aquifer at Beach Point appears to be an isolated part of the surficial aquifer identified throughout the Canal Creek area by Oliveros and Vroblesky (1989). The surficial aquifer is hydraulically interconnected to Kings Creek, Bush River, and associated wetlands. Limited studies by USGS in 1988 and 1989 showed that the surficial aquifer was tidally influenced (Oliveros and Vroblesky, 1989). Quarterly groundwater elevation monitoring by USGS in wells installed at Beach Point showed that relatively constant water levels (within tidal variations) occurred over the year, with little seasonal variation (Oliveros and Vroblesky, 1989). According to JEG (1993), this indicates that the surficial aquifer at Beach Point is more strongly influenced by tidal conditions than by recharge from infiltrating precipitation.

A recent study of the six monitoring wells described in Section 3.3 above for Beach Point confirmed that the groundwater is tidally influenced (K-V Associates, Inc., 1994). The study showed that all wells exhibited tidal influence. There were varying amounts of tidal influence including both horizontal and vertical components in the groundwater. Wells with shallower screens (CC-32A, CC-33A, and CC-34A) exhibited smaller fluctuations relative to deeper wells with a more permeable stratum (CC-32B, CC-33B, and CC-33B.1). Groundwater flow directions and rates were also substantially influenced by tidal stage. The predominant direction of horizontal flow was to the northeast.

3.6 Surface Water Hydrology

Kings Creek, which drains ≈324 ha (≈800 acres), is essentially a tidal estuary associated with the Bush River. As discussed by JEG (1993), flow from the creek appears to occur mainly as a result of tidal flushing. Net adjective flow resulting from the stream gradient appears to be minimal. Drainage into the main body of the creek is through numerous subsidiary streams and wetlands. The tidal range for the creek is typically <0.3 m (1 ft). Salinity generally varies from approximately 1 to 3 parts per thousand (ppt). The depth of the creek is <3 m (10 ft).

The Bush River, a tributary estuary of the Chesapeake Bay, has a tidal range of 0.15 to 0.46 m (0.5 to 1.5 ft) at Beach Point (JEG, 1993). The river is approximately 1.6 km (0.87 nautical mile) wide at Beach Point and is generally <1.8 m (6 ft) deep except in the shipping channel, where the depth is about 6.1 m (20 ft) (JEG, 1993). The salinity of the river in the vicinity of Beach Point can vary considerably on a seasonal basis; the average is ≈3 ppt (Pritchard, 1976).

As discussed by Carter (1976), the Bush River is ≈15.7 km (8.5 nautical miles) long measured along its axis from mouth to head. The depth increases generally from head to mouth with the maximum depth of 9.8 m (32 ft) near Briery Point; the mean depth is ≈1.8 m (6 ft). At its widest point, the river is ≈3.7 km (2 nautical miles) wide. The tide in the Bush River is semidiurnal with a mean range of 0.4 m (1.4 ft) reported for Pond Point. This tidal range results in an intertidal volume of ≈ 1.2×10^7 m^3 (42.9×10^7 ft^3). This is about 20% of the volume below mean tide level. At the time of maximum tidal velocity at the entrance, the discharge through the mouth of the river is ≈ 850 m^3 sec^{-1} (30×10^3 ft^3 sec^{-1}). The major tributaries to the river include Otter Point Creek, Lauderick Creek, and Kings Creek. The drainage of the Bush River is ≈29.5×10^3 ha (113.9 $miles^2$), resulting in a mean annual runoff of the order of 3.68 m^3 sec^{-1} (130 ft^3 sec^{-1}) (Carter, 1976). Thus, waters in the Bush are derived mainly from the adjacent Chesapeake Bay.

SECTION 4

MATERIALS AND METHODS

4.1 Biomonitoring Evaluation

Groundwater was withdrawn from well CC-33B (Harford County Permit No. HA-81-4049) which is the most highly contaminated well at Beach Point (Section 3.3). The well is 21.3 m deep (70 feet) and has a screened interval of 18.0-20.4 m (62-67 ft). The well pump intake was located at 19.8 m (65 feet). Groundwater was pumped continuously from the well at a rate of ≈7.5 L/min (2 gal/min).

Several components of the biomonitoring study were conducted on-site (see below) in an aquatic biomonitoring trailer with bioassay capabilities similar to a USABRDL trailer described by Herriott and Burton (1992). Groundwater and dilution water (APG-EA potable water) were supplied to the trailer via polyethylene pipe. The dilution water was charcoal filtered and aerated before use. Excess groundwater and diluent water from the trailer were collected, treated via charcoal, and pumped to the APG-EA Wastewater Treatment Plant for further treatment.

An array of biomonitoring systems were evaluated during the 9-month study. The biomonitoring systems included a number of endpoints. A summary of the biomonitoring tests conducted is given in Table 1. The pH of the groundwater from well CC-33B was 4 (± 0.1); thus, many of the assays were conducted at both pH 4 and pH 7. The toxicity at pH 7 was studied so that the data could ultimately be used in a hazard assessment of the groundwater as it enters the Bush River which generally has a pH in the low neutral range.

The experimental procedures and frequency of each assay are described in detail below. The following is a brief description of the tier of biomonitoring systems employed in the evaluation. Acute toxicity of the groundwater was evaluated three times each week using the 5- and 15-min Microtox® assay which uses microbial (Photobacterium phosphoreum) bioluminescent activity. In addition to providing rapid toxicity data, the test was also conducted to monitor the toxicity of the groundwater over time. A rapid 24-h rotifer (Branchious rubens) test was conducted to determine acute toxicity. The rotifer bioassay was conducted four times on a bimonthly basis during the course of the study at a pH of 4 and 7. Acute toxicity data were also obtained for the algal, invertebrate, and fish used in the short-term toxicity tests described in Section 4.1.2 below. An acute toxicity test was also conducted with the Japanese medaka (Oryzias latipes).

TABLE 1. SUMMARY OF THE BIOMONITORING TESTS CONDUCTED

Test and/or Species	Type of Test	Test No.	Test Periods	Comments
Microtox® (bacterium)	5- and 15-min EC50	N/A	03/24/93 - 12/06/93	Grab and/or composite sample[a]
Rotifer	24-h LC50	1	04/10/93 - 04/12/93	Grab and composite sample[b]
		2	06/02/93 - 06/04/93	
		3	08/24/93 - 08/26/93	
		4	12/01/93 - 12/03/93	
Green alga	96-h EC50; 96-h Growth	1	04/10/93 - 04/14/93	Grab and composite sample[b]
		2	06/03/93 - 06/07/93	
		3	08/24/93 - 08/28/93	
		4	12/01/93 - 12/05/93	
Cladoceran	48-h LC50; 7-d Survival and reproduction	1	04/07/93 - 04/14/93	Grab and composite sample[c]
		2	06/01/93 - 06/08/93	
		3	08/24/93 - 08/31/93	
		4	11/30/93 - 12/07/93	
Fathead minnow	96-h LC50; 7-d Survival and growth	1	04/07/93 - 04/14/93	Grab and composite sample[c]
		2	06/02/93 - 06/09/93	
		3	08/24/93 - 08/31/93	
		4	11/30/93 - 12/07/93	
Japanese medaka	96-h LC50	N/A	11/30/93 - 12/04/93	Grab and composite sample[c]
Genotoxicity (bacterium)	Ames assay	1	04/14/93	Grab and/or composite sample[d,e]
		2	06/02/93	
		3	08/23/93	
		4	11/30/93	

TABLE 1. (CONTINUED)

Test and/or Species	Type of Test	Test No.	Test Periods	Comments
Genotoxicity (Chinese hamster ovary cells)	SCE assay	N/A N/A N/A N/A	06/02/93 07/19/93 08/23/93 11/30/93	Grab and/or composite sample[f]
Developmental toxicity (African clawed frog)	4-d FETAX	1 2 3 4	04/10/93 - 04/14/93 06/03/93 - 06/07/93 08/26/93 - 08/30/93 12/03/93 - 12/07/93	Grab and composite sample[c]
Chronic histopathology and growth (Japanese medaka)	6 and 9 Months	N/A	02/25/93 - 11/29/93	Flow-through exposure conducted in biomonitoring lab
Comprehensive chemical analyses	N/A	1 2 3 4	04/14/93 06/02/93 08/23/93 11/30/93	Grab and composite sample[g]
Munitions analyses	N/A	1 2 3 4	04/14/93 06/02/93 09/30/93 11/29/93	Grab sample of 100% groundwater
Routine water quality analyses	N/A	N/A	Daily/weekly	Grab sample[h]

[a] Grab sample of 100% groundwater and composite grab samples of 10 and 1% groundwater by volume taken 3 times a week; composite grab sample of diluent water taken once a week.

TABLE 1. (CONTINUED)

Test and/or Species	Type of Test	Test No.	Test Periods	Comments

[b] Grab sample of 100% groundwater; composite grab sample of diluent water.
[c] Grab sample of 100% groundwater and composite grab sample of diluent water renewed every 24 h.
[d] Composite grab samples of 10 and 1% groundwater by volume and diluent water taken for analysis on 04/14/93 and 06/02/93.
[e] Grab sample of 100% groundwater and composite grab samples of 10% groundwater by volume and diluent water taken for analysis on 08/23/93 and 11/30/93.
[f] Samples taken for analyses were as follows: composite grab sample of 10% groundwater by volume on 06/02/93, grab sample of 100% groundwater on 07/19/93 and 11/30/93, and composite grab sample of diluent water on 08/23/93.
[g] Grab sample of 100% groundwater and composite grab samples of 10 and 1% groundwater by volume and diluent water taken for analysis during each test period listed.
[h] Grab samples taken from chronic histopathology tanks as described in Section 4.1.6.2.

The following 4- to 7-d short-term toxicity tests, which were used to estimate chronic toxicity, were performed on a bimonthly basis: 96-h algal (Selenastrum capricornutum) growth test; 7-d cladoceran (Ceriodaphnia dubia) survival and growth test; and 7-d fathead minnow (Pimephales promelas) survival and growth test. Four bimonthly tests were conducted with each species at pH 4 and pH 7. In addition to the short-term methods used to estimate chronic toxicity, growth data at 6 and 9 months from the chronic Japanese medaka histopathology test described below were also used as chronic toxicity endpoints.

Gene mutation potential was determined using the Ames Salmonella/mammalian-microsome reverse assay. Primary DNA damage was evaluated by the sister chromatid exchange (SCE) assay. Developmental toxicity was determined by the 96-h frog embryo teratogenesis assay-Xenopus (FETAX) using the African clawed frog, Xenopus laevis. Genotoxicity and developmental toxicity assays were conducted at bimonthly intervals during the same periods as the above acute and short-term chronic tests were conducted. Chronic histopathological changes were evaluated using the Japanese medaka as the experimental model. Both unexposed and fry exposed to diethynitrosamine (DEN) were exposed continuously under flow-through test conditions for 6- and 9-month exposure periods.

Comprehensive chemical analyses of the raw groundwater, test dilutions in the chronic histopathology assay, and diluent water were performed four times at bimonthly intervals. The chemical analyses were conducted during the same periods that the above bimonthly assays were conducted. Routine water quality analyses were also conducted at various frequencies on a weekly basis.

4.1.1 Acute Toxicity Tests

4.1.1.1 Microtox®

The Microtox® test (Microbics Corp., Carlsbad, CA) is a rapid acute toxicity test that may be completed in less than one hour. The test is based on the reduction in bioluminescence of the marine bacterium P. phosphoreum when exposed to a sample of unknown toxicity. The degree of light reduction, an indication of metabolic inhibition in the test preparation, indicates the degree of toxicity of the sample. The Microtox® test procedures followed were those outlined in Herriott and Burton (1992) which were derived from Microtox®'s operating manual (Microtox®, 1988). A Microtox® Model 500 Analyzer with PC version 6.3 software was used for both a 5-min and 15-min test on all samples.

Microtox® assays were initiated on April 2, 1993. The assays were conducted on-site three times a week until the biomonitoring study was completed on December 6, 1993. Both 5- and 15-min assays were conducted on 100, 10, and 1% groundwater

by volume samples as well as APG-EA diluent water. The 10 and 1% groundwater by volume samples were taken from the chronic histopathology study tanks as described below in Section 4.1.5. A 5- and 15-min assay was conducted once each week on 100% groundwater adjusted to pH 7 with 10 N NaOH.

4.1.1.2 Rotifer

The 24-h freshwater rotifer Toxkit™ screening test (Bio-Response Systems, Halifax, Nova Scotia) was used to determine the potential toxicity of the groundwater at pH 4 and 7 (groundwater buffered with 10 N NaOH). pH-adjusted aliquots of the same groundwater used for the short-term chronic toxicity tests were used for the rotifer assays (see Section 4.1.2). The test utilized newly hatched rotifers (B. rubens) <4 h old. The rotifers used in the tests were hatched from cysts supplied in the Rotifer ToxKit™. Rotifer ToxKit™ synthetic medium was used to hatch the cysts, rear the organisms before testing, and as diluent media for the tests. The static tests were conducted in glass Petri dishes containing 10 mL of test solution.

The groundwater, which was used within 6 h from the time of collection for the rotifer bioassays, was held in glass containers at 4°C until used in the tests. A geometric series of five groundwater concentrations (plus controls) was used. Three replicates of 10 organisms each were used at each test concentration. All tests were conducted at 25 ± 0.5°C. Routine water quality (alkalinity, conductivity, DO, hardness, pH, and temperature) was measured at the beginning and end of each test. The methods used for the chemical analyses are given in Section 4.1.6.2. All tests were conducted under a 16-h light:8-h dark photoperiod (fluorescent lights at 60-85 foot candles).

4.1.1.3 Green Alga, Cladoceran, Fathead Minnow, and Japanese Medaka

Acute toxicity values were calculated where possible at pH 4 and 7 for the green alga, cladoceran, and fathead minnow from the data obtained during the short-term chronic tests described in Section 4.1.2. With regard to the green alga, EPA's Office of Research and Development considers the 96-h algal test for growth to be a short-term chronic test for determining the toxicity of effluents (Horning and Weber, 1985; Weber et al., 1989) as do other investigators for evaluating single chemicals (for ex., see Hughes et al., 1988 and Suter, 1993). EPA's Toxic Substance Control Act office considers the 96-h test to be an acute test (U.S. EPA, 1985a and 1986a). Because we used the short-term chronic method (Section 4.1.2.1), we analyzed the data as chronic data; however, we also analyzed and reported the results as 96-h acute data so that acute:chronic ratios could be calculated for use in the hazard assessment (Section 4.2).

Forty-eight-h LC50s and 96-h LC50s were determined where possible for the cladoceran and fathead minnow, respectively. One 96-h acute toxicity test was conducted with the Japanese medaka using the procedure of Weber (1991). The age of the Japanese medaka used in the test was 22 d old at the start of the exposure; the test procedure recommends fish between the ages of 1-14 d old be used.

4.1.2 Short-term Chronic Toxicity Tests

The specific test methods for the short-term chronic tests are given below. Deviations from the test methods are discussed where appropriate. A geometric series of five groundwater concentrations (plus controls) was used in most tests. The groundwater samples used in all tests were obtained daily and used within 6 h at each 24-h renewal (see below). All groundwater samples were transported in glass containers on ice and held at 4°C until used for the tests. Each groundwater sample was split into two aliquots. One aliquot was maintained at pH 4 and the second buffered to pH 7 (10 N NaOH). All bimonthly short-term chronic bioassays were conducted at pH 4 and 7. The same pH-adjusted aliquots were also used for the rotifer bioassays (Section 4.1.1.2) and FETAX assays (4.1.4). The bioassays were conducted at the University of Maryland Wye Research and Education Center (UMD/WREC) Aquatic Toxicology Laboratory.

4.1.2.1 Green Alga

The short-term chronic toxicity of the groundwater to the green alga (S. capricornutum) at pH 4 and 7 (10 N NaOH) was determined four times by the EPA procedures given in Weber et al. (1989). A starter culture of S. capricornutum was obtained from the culture collection at the University of North Texas, Denton, TX. Stock algal cultures were reared in 2.5 L Pyrex culture flasks containing 1 L of sterilized double strength "AAP" algal assay medium, with sufficient P added to achieve a 20:1 N:P ratio as described in Miller et al. (1978). Cultures were maintained in a constant temperature incubator under constant cool-white fluorescent lights (≈300 foot candles) at a temperature of 25 ± 0.2°C on a shaker table oscillating at 100 rpm (± 10 %). Log growth cells were used to start all tests.

Algal test solutions were prepared by dilution of the groundwater with filtered sterilized assay media. Test solutions (100 mL total volume) were dispensed into 250 mL Delong flasks and inoculated with S. capricornutum cells in log growth to achieve a density of $\approx 1 \times 10^4$ cells/mL. Triplicates were prepared for each treatment. The flasks were placed on a shaker table in an incubator set at the culturing conditions described above. Growth measurements (cell density) were made from all replicates in each treatment at 0, 24, 48, 72, and 96 h. Algal

cell density was determined from a 1 mL sample with a Model ZBI Coulter Counter (Coulter Electronics, Inc., Hialeah, FL). The instrument was calibrated with each use via hemocytometer counts. Test solutions were not renewed during the 96-h studies.

4.1.2.2 Cladoceran

The chronic toxicity of the groundwater at pH 4 and 7 to C. dubia was determined four times by the EPA static renewal method (solutions renewed daily) given in Weber et al. (1989). The cladoceran was cultured at 25 ± 1°C in 600 mL glass beakers filled with 400 mL of 20% Perrier:80% reverse osmosis water amended with selenium (2 µg Se/L as Na_2SeO_3) as recommended by Winner (1989). The diet consisted of a mixture of Cerophyl® (Cerophyl Laboratories, Inc., Kansas City, MO) and the green alga, S. capricornutum, added to the cladoceran culture to achieve final concentrations of 120 µg Cerophyl®/mL and 6.7×10^5 S. capricornutum cells/mL. Starter cultures of C. dubia were obtained from the Center for Lake Superior Environmental Studies, University of Wisconsin - Superior.

All neonates used in the 7-d survival and reproduction tests were produced by cladocerans in culture that had released at least three broods. The initial age of the neonates in each test was <12 h old. The tests were conducted in 50 mL glass beakers containing 30 mL of test solution. All tests were conducted in an environmental chamber at 25 ± 1°C under a 16-h light:8-h dark photoperiod (fluorescent lights; 60-85 foot candles at the surface of the culture vessels). All test organisms were fed daily as described above at each 24-h renewal. Routine water quality was taken at the beginning and end of each 24-h renewal. The methods used for the chemical analyses are discussed in Section 4.1.6.2.

4.1.2.3 Fathead Minnow

The toxicity of the groundwater at pH 4 and 7 to fathead minnows (P. promelas) was determined four times by the EPA static renewal method (solutions renewed daily) given in Weber et al. (1989). All larvae used in the 7-d survival and growth tests were <24 h old at the start of the test. The tests were conducted in 600 mL glass beakers containing 400 mL of test solution. The dilution water was a 20% Perrier:80% reverse osmosis water. All test organisms were fed brine shrimp (Artemia sp.) nauplii <24 h old daily at each 24-h renewal. All tests were conducted at 25 ± 1°C under a 16-h light:8-h dark photoperiod (fluorescent lights; 60-85 foot candles). Routine water chemistry was performed at the beginning and end of each renewal. Dry weight was determined by drying at 100°C for a minimum of 12 h.

Fathead minnow larvae were obtained from the UMD/WREC culture maintained at 25 ± 1°C in UMD/WREC non-chlorinated well water (mean dissolved oxygen = 8.2; pH = 7.8; conductivity = 161 µs/cm; alkalinity = 53 mg/L as $CaCO_3$; hardness = 52 mg/L as $CaCO_3$). The UMD/WREC culture procedures were similar to those recommended by Peltier and Weber (1985). The UMD/WREC culture was initiated with mature fathead minnows obtained from the U.S. EPA Environmental Monitoring and Support Laboratory - Cincinnati, Ohio.

Spawning fish were cultured in fiberglass tanks (2.4 x 0.8 x 0.5 m) containing 0.2 m UMD/WREC well water held at 25 ± 1°C. The spawning adults were fed a diet of frozen brine shrimp (Artemia sp.; Argent Chem. Lab., Redmond, WA) and TetraMin® Staple Food (Ramfab Aquarium Products Co., Oak Ridge, TN) twice daily. Excess food was removed daily. Four sets of spawning fathead minnows were maintained in the culture tanks at a ratio of 1 male:4 females. Replacement spawners were rotated at approximately 3-month intervals. Fathead minnow embryos were collected on spawning substrates (10 cm I.D. x 20 cm long PVC pipe sections cut longitudinally in equal portions) and transferred to 19 L aquaria at 25 ± 1°C in UMD/WREC well water for hatching. All stages of the fish were reared under a 16-h light:8-h dark photoperiod (fluorescent lights; 60-85 foot candles).

4.1.3 Genotoxicity Tests

4.1.3.1 Gene Mutation Assay

The Ames assay was used to predict chemical mutagenic activity which in turn may serve as a carcinogen prescreen test (Ames et al., 1973). Salmonella typhimurium/mammalian-microsome reverse mutation assays were conducted four times on the groundwater and APG diluent water samples described below. The assays were conducted on both unconcentrated and concentrated (10X via XAD-2 resin extracts) samples of the groundwater and diluent water. The Ames mutagenicity assays were conducted by Hazleton Washington, Inc., Vienna, VA.

Grab samples of groundwater were collected in polypropylene containers as follows. Raw groundwater was taken directly from the well fed line to the biomonitoring laboratory. Groundwater samples of 10 and 1% groundwater by volume were taken as composite samples from the chronic histopathology fish tanks described in Section 4.1.5. Grab samples of APG-EA diluent water were taken from a large polypropylene tank with a 99% particle replacement time of ≈12 h. Thirty-one liters (1 L for an unconcentrated sample and 30 L for a 10X sample) of each material were siphoned into Nalgene polycarbonate carboys, packed in ice, and transported to Hazleton Washington, Inc. in insulated containers. The unconcentrated samples were analyzed by Hazleton

Washington, Inc. Protocol No. HLA Protocol 401W, Edition 17. The concentrated (10X) samples were analyzed by Protocol No. HLA Protocol 401X, Edition 18.

Detailed experimental procedures for the unconcentrated and 10X tests are given in the protocols listed above. Briefly, the mutagenicity assays evaluated the groundwater and diluent water samples for their ability to induce reverse mutations at the histidine locus in the genome of specific S. typhimurium tester strains both in the presence and absence of an exogenous metabolic activation system of mammalian microsomal enzymes derived from Aroclor 1254-induced rat liver. The tester strains used in the assays were TA98 and TA100. The assays were conducted using two plates per dose level in the presence of microsomal enzymes. Six dose levels of the groundwater and diluent water samples were tested in both the presence and absence of S9 along with appropriate vehicle controls (three plates per dose), negative controls, and positive controls. Resin controls were also run for the 10X samples. The doses tested in the 10X assays varied based on the amount of extractable organics recovered from the test material.

4.1.3.2 Sister Chromatid Exchange Assay

The sister chromatid exchange (SCE) assay was used as a test for possible genetic damage induced by chemical mutagens (Perry and Evans, 1975). We are aware that many mutagens are active in SCE assays, yet, it is still incompletely understood how DNA damage or perturbations of DNA synthesis give rise to SCE (Hoffman, 1991). The assay was performed once on concentrated samples (≈50,000X) of 100% raw groundwater, 10% groundwater by volume, and APG-EA diluent water. One assay was also conducted on unconcentrated 100% groundwater. The SCE assays were conducted by Hazleton Washington, Inc., Vienna, VA.

Grab samples of groundwater were collected in polypropylene containers as follows. The 100% groundwater samples were taken directly from the well fed line to the biomonitoring laboratory. The 10% groundwater by volume sample was taken as a composite sample from the chronic histopathology fish tanks described below. A grab sample of APG-EA diluent water was taken from a large polypropylene tank with a 99% particle replacement time of ≈12 h. Thirty liters were siphoned into Nalgene polycarbonate carboys, packed in ice, and transported to Hazleton Washington, Inc. in insulated containers. The samples were assayed by Hazleton Washington, Inc. using their Genetics Protocol No. 438, Edition 15.

The experimental procedures for the SCE assay are given in the protocol listed above. Briefly, 30 L of each sample were concentrated in DMSO to a final volume of 6 mL (≈50,000X) by rotoevaporation. The unconcentrated 100% groundwater sample was

not concentrated by rotoevaporation. The extract was evaluated for its ability to induce sister chromatid exchanges (determined at metaphase) in Chinese hamster ovary (CHO) cells in the presence and absence of an exogenous metabolic activation system of mammalian microsomal enzymes derived from Aroclor 1254-induced rat liver. The SCE assays were conducted by exposing cultured cells to the test extract, growing the cells with the thymidine analog of 5-bromo-2'-deoxyuridine for about two cell cycles, and making chromosome preparations that were Giemsa-stained for SCE. The doses tested were selected based on the amount of extractable organics recovered from the test article. Single cultures of CHO cells were incubated with five concentrations of the extract with and without metabolic activation. Sister chromatid exchange frequencies were analyzed in the cultures treated with the two highest doses with second generation cells and from solvent and positive control cultures.

4.1.4 Developmental Toxicity Test

Four bimonthly developmental toxicity tests were conducted at pH 4 and 7 using the frog embryo teratogenesis assay - Xenopus (FETAX). The assay is a 96-h quantitative developmental assay used to screen for developmental toxicants in aquatic media. The assays were conducted using the static renewal (solutions renewed every 24 h) test method Designation E 1439-91 of the American Society for Testing and Materials (ASTM, 1992a). Embryolethality and malformations were determined; growth retardation was not evaluated. The identification and interpretation of malformations in the embryos at 96 h were made via the atlas of Bantle et al. (1991). Aliquots of the same groundwater used for the rotifer, algal, and short-term chronic toxicity biomonitoring tests were used for the FETAX assays (Section 4.1.2).

Embryos between normal stage 8 blastulae and normal stage 11 gastrulae were obtained from X. laevis breeding colonies at the UMD/WREC as described below. The embryos were de-jellied in a 2% L-cysteine solution (2 g of L-cysteine per 98 mL of FETAX solution). Once de-jellied, the embryos were rinsed and re-suspended in FETAX solution (ASTM, 1992a). The embryos were tested in glass petri dishes containing 10 mL of solution. Two replicates of 25 embryos/replicate were used for each test treatment. As required by the study protocol, four replicates of 25 embryos/replicate were used for the control treatment. The tests were conducted at 24 ± 0.2°C under a 16-h light: 8-h dark photoperiod (fluorescent lights; ≈75 foot candles at the surface of the test medium) in a constant temperature environmental chamber.

The UMD/WREC X. laevis adult colony was maintained in flow-through (≈4 replacement volumes per day) circular polyethylene aquaria (0.91 m I.D. x 0.36 m high) with a water depth of 10 cm. Each aquarium contained a maximum of 10 adults. UMD/WREC non-

chlorinated deep well water (water quality given in Section 4.1.2.3) held at 23.5 ± 0.5 °C served as the culture medium. All frogs were fed every 5-6 d with commercial beef liver supplemented with liquid vitamins (PolyViSol; Mead-Johnson Nutritionals, Evansville, IN). The colony was held under a photoperiod of 16h light:8 h dark. Mating pairs were bred in the dark in 23.5 ± 0.5 °C UMD/WREC non-chlorinated water at ≈70 d intervals by injecting 400 and 800 I.U. of human chorionic gonadotropin (HCG) in the dorsal lymph sac of the males and females, respectively. Amplexus occurred 4-6 h after injecting HCG; egg deposition occurred 9-12 hours following HCG injection. The original breeding stock was obtained from Xenopus I (Ann Arbor, MI).

4.1.5 Chronic Histopathology and Growth Test

Chronic histopathologic changes were evaluated using the Japanese medaka (O. latipes) as the experimental model. The Japanese medaka is a sensitive laboratory model for screening environmental pollutants which may induce histopathological changes and neoplasms (for ex., see Hawkins et al., 1988; Klaunig et al., 1984; Metcalfe, 1989). Both unexposed and fry exposed to diethylnitrosamine (DEN) were exposed continually under flow-through test conditions for a 9-month period. A subset of organisms was taken after 6 months of exposure for morphometric measurements and histopathological evaluation. The USABRDL test designation was Protocol No. 401-001R.

The fish were exposed to one of three treatments: 10% groundwater by volume, 1% groundwater by volume, or APG-EA diluent water (control). A 100% groundwater treatment could not be used because the pH of the groundwater was 4 (±0.1) which would have caused excessive mortality over the 9-month exposure period. Consideration was given to buffering the 100% groundwater to pH 7 and conducting the 100% treatment at pH 7. However, a preliminary evaluation of the buffered groundwater at pH 7 showed that excessive precipitation of metals occurred. Thus, 100% groundwater was not buffered to pH 7 and used as an experimental treatment.

The flow-through test solutions were delivered by a solenoid-activated proportional dilutor system which was constructed primarily of glass and stainless steel; some silicon tubing was also used. The test concentrations were delivered to twelve 19 L (5 gal) glass aquaria (4 aquaria at 10% groundwater by volume; 4 at 1% groundwater by volume; and 4 control aquaria); each aquarium contained a volume of ≈16 L (4.25 gal). All aquaria were held at 25 ± 1°C in a constant temperature water bath. The dilutor was calibrated to complete one full cycle every 2.5 to 3.5 min. During a cycle, tanks 1-4 each received 300 ± 15 mL of APG-EA diluent water, tanks 5-8 each received 300 ± 15 mL of 10% groundwater by volume, and tanks 9-12 each

received 300 ± 15 mL of 1% groundwater by volume.

Both unexposed fry and fry exposed to DEN, were reared off-site at USABRDL until 17 d old. The DEN-initiated fish were exposed to 10 mg/L DEN for 48 h when the organisms were 12 to 14-d old. Prior to the start of the exposure to groundwater, the 17-d old fish were randomized into 6 groups of 60 fish/group for both the unexposed and DEN-initiated groups. The fish were suspended in 12 1-L mesh-bottom glass beakers in the appropriate flow-through test aquaria in the biomonitoring laboratory. The fish were held in the beakers for one week after which they were released into the aquaria.

Japanese medaka, 17-22 d old, were fed microworms two feedings per day and live brine shrimp (Artemia sp.) (<24 h old) two feedings per day (30 brine shrimp/fish). Pre-adult fish, 23-30 d old, were fed Tetramin® flake food two feedings per day and live brine shrimp <24 h old (one feeding per day; 40 brine shrimp per fish). Adult fish, >30 d old, were fed Tetramin® flake food (three feedings per day on Tuesday and Thursday and two feedings per day on the remaining days) and live brine shrimp (one feeding per day on Monday, Wednesday, and Friday). The ration was adjusted as the size of the fish increased. Tanks were cleaned on an as needed basis (usually 1-2 times a week) by scrubbing algae from the sides of the tanks, allowing the debris to settle, and then siphoning. Tetramin® was fed ad libitum for 15-30 min during each feeding.

The number of test organisms alive in each tank were monitored and recorded daily. Moribund fish were euthanized and fixed in Bouin's solution for subsequent histological observation. The dilutor cycle time was calculated and recorded daily. The volume of groundwater and diluent water delivered to the aquaria were checked weekly. When necessary, cycle time and/or volume distributions were adjusted. The dilutor was occasionally shutdown (for no more than one hour) and cleaned on an as needed basis. Daily water quality (DO, pH, and temperature) was determined in all aquaria. Additional water quality tests (alkalinity, hardness, conductivity, total residual chlorine, free available chlorine, and total ammonia-nitrogen) were performed once a week in all aquaria (Section 4.1.6.2). A 16-h light:8-h dark photoperiod (fluorescent lights at 70-100 foot candles) was maintained throughout the study. Unionized ammonia-nitrogen was determined by the method of Thurston et al. (1979). Comprehensive chemical analyses were performed four times at bimonthly intervals as discussed in Section 4.1.6.1 on 100, 10, and 1% groundwater by volume and APG-EA diluent water during the test periods shown in Table 1.

On day 180, 20 Japanese medaka in each tank were removed and taken back to USABRDL for fixation (Bouin's solution) and subsequent histological observation. Wet weight and standard

length measurements were taken on all fish. The morphometric data were taken to assess the effects of a chronic 6-month exposure to the contaminated groundwater as well as the general health of the fish. On day 278, when the exposure was completed, the remaining Japanese medaka were also taken back to USABRDL for morphometric measurements and subsequent histological analysis. The histological analyses for the 6- and 9-month exposures were performed by Experimental Pathology Laboratories, Inc., Herndon, VA.

4.1.6 Chemical Analyses

4.1.6.1 Comprehensive Chemical Analyses

Comprehensive chemical analyses were performed four times at bimonthly intervals on 100, 10, and 1% groundwater by volume and APG-EA diluent water during the test periods shown in Table 1. The comprehensive chemical analyses included general water quality, metals, volatile organics, acid compounds, base/neutral compounds, pesticides, herbicides, organophosphorus pesticides, and munitions. The elements and/or compounds analyzed in each group are presented in the data tables discussed in Section 5.1.6.1. The 100% groundwater samples were grab samples taken directly from the well fed to the biomonitoring trailer. The 10 and 1% groundwater by volume samples were composite samples taken from the four 10 and 1% groundwater by volume replicate treatment tanks in the chronic histopathology study (Section 4.1.5). Grab samples of APG-EA diluent water were taken from a large polypropylene tank with a 99% particle replacement time of ≈12 h.

The groundwater and APG-EA diluent water samples were placed in appropriate containers for various analyses. The containers were placed on ice and delivered to a vendor for the analyses. The first comprehensive analysis was performed by Biospherics Inc., Laurel, MD, for all materials with the exception of the munitions. The remainder of the analyses were performed by Gascoyne Inc., Baltimore, MD, for all materials with the exception of the munitions. The methods used for the analyses of all materials are given in the data tables discussed in Section 5.1.6.1. The four munitions samples were analyzed by USABRDL via in-house procedures (U.S. Biomedical Research and Development Laboratory, 1993).

4.1.6.2 Routine Water Quality Analyses

Routine water quality was measured in all histopathology treatment tanks. Dissolved oxygen, pH, and temperature were measured daily. Alkalinity, hardness, conductivity, total residual chlorine, free available chlorine, and total ammonia-nitrogen were measured once a week (all tests were performed on the same days). Unionized ammonia-nitrogen was determined by the method of Thurston et al. (1979). The methods used for the

analyses are presented in Section 5.1.6.2. The same methods used in the biomonitoring trailer were also used for the toxicity studies conducted at the University of Maryland Wye Research and Education Center.

In addition to the temperature measurements made in the aquaria during the chronic histopathology test, temperature was monitored continuously in Tank No. 3 via a strip chart recorder (Cole-Parmer Thermistor Recorder Model No. 08354-15, Cole-Palmer Instrument Co., Chicago, IL).

4.1.7 Test Endpoints and Data Analyses

The test endpoint for the Microtox® 5- and 15-min EC50s was a reduction in bioluminescence. The EC50s and their 95% fiducial limits were determined by probit analysis using the software program supplied by Microtox® (Microtox®, 1988). The test endpoint for the acute effects of groundwater to the green alga was growth, measured as density (cells/mL). The 96-h EC50s for growth were estimated by using the "inhibition proportion" technique recommended by Horning and Weber (1985). The technique uses quantal analyses (e.g., probit or moving average angle methods) to estimate EC50s and their 95% fiducial or confidence limits. Since the assumptions of the quantal analysis are not met in the classical sense because of the very nature of the growth data, the count data at each treatment were averaged and subsequently converted to "inhibition proportions" using the formula below before a moving average angle analysis was performed (Stephan, 1978).

$$I = C - T / C * 100$$

where: C = the mean growth of the controls
T = the mean growth at a given treatment

The 96-h EC50s and their 95% confidence limits for embryo malformations in the FETAX assays were determined by the moving average angle method using an EPA statistical program (Stephan, 1978). The test endpoint for all 24-h LC50 toxicity tests with rotifers, 48-h and 7-d LC50 tests with cladocerans, and 96-h and 7-d LC50 tests with fathead minnows was mortality. The LC50s and their 95% confidence limits were determined by the moving average angle method (Stephan, 1978).

The test endpoint for the chronic toxicity of groundwater at pH 4 and 7 to the green alga was growth measured as density (cells/mL). The no-observed-effect concentrations (NOEC) and lowest-observed-effect concentrations (LOEC) were determined by Dunnett's test. Dunnett's test consists of an analysis of variance (ANOVA) to determine the error term, which is then used in a multiple comparison test for comparing each of the treatment means with the control mean. The assumptions upon which the use

of Dunnett's test are contingent are that the observations within treatments are independent and normally distributed, with homogeneity of variance. The chi-square test for normality and Bartlett's test for homogeneity of variances were performed before the Dunnett's test was used. The above statistical tests were perform using Toxstat (Gulley et al., 1989) at a minimum probability level of 0.05.

The endpoints for the 7-d survival and reproduction tests with Ceriodaphnia were survival and young production. The endpoints for the fathead minnow 7-d survival and growth tests were survival and growth. The endpoints for the 96-h FETAX assay were survival and number of malformations. The statistics used for the LC50 data and FETAX EC50 (malformations) data are given above. NOECs and LOECs were determined as follows. The adult raw cladoceran survival data were analyzed by Fisher's Exact test. Arc-sine square root transformations were made on the FETAX percent embryo survival and percent embryo malformation data as well as the fathead minnow percent survival raw data before further data analyses were performed. With the exception of the cladoceran survival data, all data were then subjected to a chi-square test of normality and Bartlett's test for homogeneity of variance.

When the data sets met the assumptions of normality and homogeneity of variance, a parametric statistic was used. Dunnett's test was used when the number of replicates was constant among treatments. A t-test with Bonferroni adjustment of error rate was performed when the number of replicates was not constant among treatments. When a data set failed to meet the assumptions of normality or homogeneity of variance, a nonparametric statistic was used. Steel's Many-One Rank test was performed when equal number of replicates were used. The statistical tests were performed using Toxstat (Gulley et al., 1989). A minimum probability level of 0.05 was used for all tests.

The morphometric endpoints for the Japanese medaka chronic histopathology study after 6 and 9 months of exposure were wet weight and standard length. The raw data were first checked for normality and homogeneity of variance by the Kolmogorov-Smirnov test and Levene median test, respectively. The wet weight data after 6 and 9 months of exposure passed the tests for normality and homogeneity of variance and were subsequently analyzed by a general linear model type III test (replicates within treatment of the error term). The standard length data after 6 and 9 months of exposure were not normally distributed; thus, the non-parametric Kruskal Wallis one way ANOVA on ranks statistic was used. A statistically significant difference ($\alpha = 0.05$) was found for length after 9 months of exposure using the nonparametric Kruskal Wallis test. Thus, Dunn's multiple comparison test was used to enumerate the difference between the

control and each experimental treatment. All statistical tests with the exception of the general linear model were performed via Sigma Stat (1992). The general linear model was run with SAS (1987). A minimum probability level of 0.05 was used for all tests. The histopathological data enumerated by Experimental Pathology Laboratory, Inc. at 6 and 9 months were not treated statistically.

4.2 Hazard Assessment Evaluation

The U.S. Army Aberdeen Proving Ground Directorate of Safety, Health, and Environment (DSHE) is conducting a risk assessment of the Canal Creek area as part of their Installation/Restoration Program. The potential hazard of the contaminated surficial aquifer at Beach Point, which appears to be an isolated part of the surficial aquifer identified throughout the Canal Creek area (Section 3.5), is being addressed by DSHE as an operable unit in a focus feasibility study of Beach Point. A hazard assessment of the contaminated surficial aquifer to aquatic organisms was performed in this study using the toxicity data obtained in the biomonitoring evaluation study. The ASTM Designation E 1023-84 (Reapproved 1988) standard guide for assessing the hazard of a material to aquatic organisms was used as the basis for the hazard assessment evaluation (ASTM, 1992b).

4.2.1 Scope of the Hazard Assessment

The hazard assessment process is complex and requires decisions at a number of points (ASTM, 1992b). The validity of a hazard assessment depends on the soundness of the decisions and the accuracy of information used. All decisions should be based on reasonable worst-case analyses so that an appropriate assessment can be completed for the least cost that is consistent with scientific validity. A number of worst-case assumptions were made for the Beach Point assessment when sufficient data were not available in order to err on the conservative side during the evaluation. The use of worst case assumptions obviously introduces a significant bias in the hazard assessment.

As discussed in the ASTM guide (ASTM, 1992b), the hazard assessment procedure is an iterative process for assessing the hazard of a material to aquatic organisms. The process basically considers the relationship between a material's measured or estimated environmental concentrations and the adverse effects likely to occur to aquatic organisms. Normally the iterative process proceeds as follows. Unavailable necessary information concerning environmental concentrations and adverse effects is obtained through a stepwise process that starts with inexpensive information gathering and progresses to more comprehensive data analyses and experimental studies if necessary. At the end of each iteration (usually a maximum of three) the estimated or measured environmental concentrations are compared with

information on possible adverse effects to determine the adequacy of the available data for assessing hazard. If it is not possible to conclude that hazard is either minimal or potentially excessive after an iteration, further iterations are conducted until the hazard is adequately characterized.

4.2.2 Deviations from the Hazard Assessment Procedures

The hazard evaluation process for Beach Point deviated from some procedures recommended in the ASTM guide (ASTM, 1992b). Most of the deviations, all of which are discussed below, were dictated by the study site and the biomonitoring evaluation study design. The following deviations occurred. The ASTM guide was written for the hazard evaluation of specific chemicals or a group of chemicals that have similar biological, chemical, physical, and toxicological properties. The contaminated groundwater at Beach Point is a multiple mixture of heavy metals and chlorinated aliphatic compounds which have very different properties. Because the groundwater contained a mixture of contaminants, the complete mixture was tested rather than individual components of the mixture.

Aquatic toxicological models currently do not exist which allow for good predictions of toxicity for contaminant mixtures that contain chemicals with different modes of toxic action. Since the toxic interactions of the contaminants could not be readily determined, the concentrations of the groundwater chemicals were expressed as percent groundwater by volume rather than individual chemical mass. The mass of each contaminant in the groundwater, however, can be calculated from the chemical analysis data obtained during the biomonitoring phase of the study if desired.

The ASTM guide recommends a phased hazard iteration process (ASTM, 1992b). Several toxicological phases, however, were conducted simultaneously during the biomonitoring evaluation. That is, low-cost, medium-cost, and high-cost toxicological data collection/iteration phases were all conducted simultaneously during the biomonitoring evaluation studies.

Phase II (medium-cost information) and Phase III (high-cost information) of the ASTM hazard assessment recommend that acute and chronic toxicity tests, respectively, be conducted with estuarine organisms when a material enters estuarine waters. The Bush River is a low salinity estuary at Beach Point (Carter, 1976); however, all of the biomonitoring studies were conducted with freshwater organisms. Tests with estuarine organisms were initially considered in the design phase of the biomonitoring study. However, the pH of the groundwater at Beach Point was 4 ±0.1. Preliminary pH adjustment studies in freshwater showed that some heavy metals were "salted out" of solution when the pH was raised from 4 to 7. The "salting out" effect observed in

freshwater would increase at a salinity of 20 ppt which is the test salinity normally required for the EPA acute and short-term chronic test methods for estuarine organisms (Weber, 1991; Weber et al., 1989). Several heavy metals in the groundwater occurred at concentrations which could be toxic (U.S. EPA, 1986b); thus, a decision was made not to conduct estuarine tests because the tests would have underestimated the toxicity of the complex chemical mixture of the contaminants. As will be shown in the Results and Discussion Section, less toxicity occurred in several of the freshwater biomonitoring tests conducted at pH 7 relative to those conducted at pH 4. Finally, the State of Maryland Code (COMAR) considers the Bush River Area (Sub-Basin 26.08.02.03-1B) as "freshwater" for the purposes of applying numerical toxic substance criteria (see Section 5.2.3.1).

One can argue that differences exist between the toxicological sensitivities of freshwater and saltwater organisms. Indeed, there is no question that examples could be given to support either side of the argument with individual materials present in the groundwater at Beach Point. EPA has developed both freshwater and saltwater numerical water quality criteria for most of the priority pollutant heavy metals present in the groundwater at Beach Point (U.S. EPA, 1986b). As one would predict from the general toxicological literature, differences exist between EPA's freshwater and saltwater acute and chronic criteria for the various metals. Little research has been conducted on the comparative sensitivities of freshwater versus saltwater organisms exposed to multiple mixtures of heavy metals and aliphatic organics. Since toxicological differences exist between various freshwater and saltwater organisms and few data are available to predict differences in multiple mixtures, we assumed that the freshwater biomonitoring organisms would serve as reasonable surrogates for estuarine organisms in the Bush River.

It is known that the groundwater enters the Bush River through the bottom sediments (Section 3.5). One could argue that benthic as well as water column organisms should also be included in the suite of test organisms. No benthic organisms were tested in the biomonitoring evaluation of the groundwater. A recent analysis of numerical water quality criteria chemicals by EPA shows that freshwater and saltwater benthic organisms, in general, have toxicological sensitivities similar to those of water column organisms (U.S. EPA, 1993a). Thus, toxicity data for water column organisms were used to predict toxicity for benthic organisms.

4.2.3 Estimated Discharge Rate of the Surficial Aquifer

The discharge rate of the Beach Point surficial aquifer to the Bush River had to be estimated in order to estimate the environmental concentrations of the groundwater in the Bush

River. An estimated discharge rate of the surficial aquifer at Beach Point was made by Jacobs Engineering Group (JEG, 1994). The horizontal transport value, V_{hor}, for the discharge calculation was taken from a 25-h study conducted in late March 1994 by K-V Associates, Inc. (1994). V_{hor}, which was 0.27 m/d (0.89 ft/d) in a direction of 61.8°NE magnetic north, approximately 24° riverward from the main peninsula axis, was taken from well CC-33B.1 which is an intermediate depth well at Beach Point (Section 3.3). The assumption was made that V_{hor} was continuous throughout the surficial aquifer at Beach Point (JEG, 1994). V_{hor} orientation (flow direction) was assumed to be consistent with actual aquifer conditions over one complete tidal cycle (discounting short-term fluctuations due to tidal changes).

The flowing assumptions were also made for the calculation of the estimated discharge rate of the surficial aquifer at Beach Point (JEG, 1994). The surficial aquifer was considered to be homogeneous, characterized by isotropic flow conditions. The average saturated thickness of the aquifer was assumed to be 18.8 m (61.8 ft). The Bush River frontage along the peninsula is ≈448 m (1,470 ft). Approximately 366 m (1,200 ft) of frontage is directed in a NE direction; approximately 82 m (270 ft) of the northern peninsula terminus is directed in a NW direction. The aquifer was assumed to discharge the entire length of the Beach Point Peninsula fronting the Bush River. Thus, the cross-sectional area of saturated surficial aquifer fronting Bush River was assumed to be ≈8,443 m^2 (90,883 ft^2). The length and area of the aquifer fronting the Bush River are believed to be quite conservative. A more realistic estimate of length of discharge from Beach Point is probably one third to one half, rather than the entire length (Lorah, 1994). The directional component of V_{hor} orthogonal to the Bush River frontage was used rather than assuming V_{hor} is orientated perpendicular (orthogonal) to the Bush River frontage in the surficial aquifer. That is, the measured horizontal velocity (0.27 m/d) was multiplied by a sin (24°) factor along the 366 m (1,200 ft) frontage.

The calculation of the estimated discharge rate of the surficial aquifer to the Bush River made by Jacobs Engineering Group (JEG, 1994) is:

Discharge = (0.27 m/d velocity)(sin 24°)(18.8 m depth)(366 m frontage) + (0.27 m/d velocity)(18.8 m depth)(82 m frontage)

= 1,175.5 m^3/d (41,508.7 ft^3/d)

4.2.4 Contaminant Distribution Assumptions in the Groundwater

As discussed in Section 3.3, chemical monitoring of the wells at Beach Point showed that the highest concentrations of heavy metals in 1988 occurred in the mid-depth (CC-33B.1) and deep (CC-33B) wells. The highest concentrations of chlorinated aliphatic organics occurred primarily in the deep well as a residual DNAPL plume. Only one shallow well (CC-33A) and one deep well (CC-33B) were studied at Beach Point during 1989. The concentrations of heavy metals and volatile organics were all higher in the deep well (CC-33B) than in the shallow well (CC-33A).

The following assumptions were made for the hazard assessment. The assumption was made for heavy metals that no retardation occurred via adsorption onto solid surfaces or trapping by clays through ion exchange. It was also assumed that no precipitation of the metals occurred when the pH shifted from 4 to neutrality when the groundwater entered the receiving stream. With regard to the DNAPLs, the assumption was made that no abiotic (chemical) or biotic (microbial) transformations occurred. We assumed that the heavy metals and residual DNAPLs were homogeneously mixed throughout the complete aquifer. In addition, the highest concentration (not the average concentration) of each contaminant measured during the four bimonthly chemical analyses (Section 4.1.6.1) was assumed to be homogeneously distributed throughout the aquifer. It was also assumed that the maximum concentrations of materials would not increase above those currently present in well CC-33B since the original sources of the contaminants were not longer present (Section 3.2). The assumption was made that the highest concentrations of the heavy metals and chlorinated aliphatics in the aquifer all moved through the sediments into the Bush River. Finally, the assumption was made that 1,176 m^3/d of contaminated groundwater moved into the Bush River. We are aware that some of the groundwater flow, which is influenced by tidal stage, may be in the direction of Kings Creek (Section 3.5).

4.2.5 Screening-level Assessment of the Near-Field Dilution of the Groundwater Plume in Bush River

A screening level assessment of the near-and far-field dilution of the groundwater discharge from Beach Point was conducted by Najarian Associates, Inc. (1994) under contract to USABRDL. The primary objective of the study was to estimate, using EPA-approved screening-level techniques, the initial ("near-field") dilution and flushing rate ("far-field" dilution) of the groundwater discharge plume into the Bush River. The assessment, when combined with the discharge concentration data, provides first-order estimates of near-field concentrations. That is, it gives receiving water concentrations that would be

expected over relatively short distances from the discharge site (e.g., ones, tens, and hundreds of meters) and over short time periods (e.g., seconds and minutes).

The second objective was to compute concentrations for selected toxic constituents and to compare the results to available baseline data. As a result, potential water quality impacts can be quantified. The third objective was to assess the applicability of a Maryland "regulatory mixing zone", i.e., a localized discharge zone in which local water quality standards may be exceeded (U.S. EPA, 1991a). Such zones are routinely allowed for wastewater treatment plant discharges. If applicable to the Beach Point groundwater discharge plume, a mixing zone may allow for local exceedences of water quality standards.

4.2.5.1 Near-field Dilution Estimates

The near-field model approach was to evaluate the volume of groundwater emanating from Beach Point as a function of the potential dilution available within the Bush River. Within the near-field, it was postulated that the groundwater plume intersects the channel of the Bush River and seeps into the River as a submerged buoyant plume. Based on the available geologic information about the site, it was assumed the discharge from the surficial aquifer was limited to a relatively narrow band that parallels the interface between the Bush River and the Beach Point Peninsular. Since the groundwater plume is less dense (less saline) than the adjacent estuarine water, it would rise and subsequently mix with the ambient receiving waters. Thus, it would be diluted through mechanisms of buoyancy-based entrainment and mixing associated with ambient currents.

As discussed in Najarian Associates, Inc. (1994), the same physical mechanisms are known to dilute effluent plumes discharged from outfall diffusers of wastewater treatment plants (WWTP) and electrical generation facilities. Thus, the dilution of groundwater emanating from Beach Point can be conceptualized as the dilution of a discharge from a line diffuser. This approach allows the near-field impact of the groundwater discharge to be simulated using established plume dilution models that have been developed primarily for such outfall diffusers.

The limitations of the approach are twofold. First, most plume dilution models also simulate momentum-based entrainment, a phenomenon that has a negligible effect upon the groundwater plume. Therefore, the port configuration on the conceptualized line diffuser must be set to minimize initial momentum. Secondly, the conceptualization of the groundwater discharge surface as a line diffuser will concentrate the plume's impact within the near-field condition. As this effect reduces the dimensions of the near-field plume and calculated levels of dilution, the model result will provide a conservative estimate

of field conditions.

A series of EPA-approved, plume dilution models for surface waters are given in Muellenhoff et al. (1985). The EPA models are usually applied to discharges from line sources such as outfall diffuser manifolds. In the present case, however, the groundwater discharge is emanating from the river bed. While all EPA-approved plume models could be run, ULINE (uniform linear density flume model) was selected for the dilution calculations because it is based directly on laboratory experiments and has proved it's usefulness in recent outfall studies conducted throughout the United States (Najarian Associates, Inc., 1994). The ULINE model also neglects initial discharge momentum which is consistent with the groundwater plume movement into the Bush River. The ULINE model is designed to give dilution ratios of a discharge in a receiving stream; not distance isopleths from the source of a discharge.

Outfall dilution studies are typically conducted under "critical" minimum dilution conditions (Najarian Associates, Inc., 1994). These conditions are defined as the lowest 10th percentile ambient currents, and highest 10th percentile ambient density stratification conditions. However, Maryland regulations require that the acute aquatic life toxicity criteria be applied under conditions of mean low water and "minimum daily averaged 1-h tidal velocity" (near slack tide) (COMAR 26.08.02.05C); chronic aquatic life criteria are applied under conditions of mean water level and average tidal velocity (COMAR 26.08.02.05D). Thus, the input data were selected for the ULINE near-field model to reflect these required conditions.

To adapt ULINE to the study area, an input of site-specific data was required. The model input data fell into three categories which included 1) groundwater discharge parameters; 2) receiving water parameters; and 3) outfall diffuser parameters. The estimation of the groundwater input parameters were those given in Section 4.2.3. The ambient receiving water input parameters (e.g., ambient current velocity, salinity, temperature, and depth of discharge in the Bush River) are given in Najarian Associates, Inc. (1994). The conceptualized outfall discharge parameters for an assumed 448-m line source of buoyancy are also given in Najarian Associates, Inc. (1994).

4.2.5.2 Far-field Dilution Estimates

The Bush River receiving waters can be divided into two zones based on physical mechanisms of mixing. In the near-field, rapid mixing is caused primarily by the discharge plumes's buoyancy and initial discharge momentum, in combination with ambient currents. In the far-field, the plume is mixed passively by ambient processes such as turbulence and tidal dispersion. Far-field length scales are typically kilometer distances from

the discharge site; far-field time scales are hours, days, or longer.

Three EPA methods were initially considered for the far-field dilution estimates (i.e., the flushing rate) in the Bush River (U.S. EPA, 1985b). Far-field dilutions were calculated and reported by Najarian Associates, Inc. (1994) for the Bush River via the three methods, which included the tidal prism method, fraction of freshwater method, and dye-tracer method. The dye-tracer method was selected for further computations because dye-tracer data collected by Carter (1976) for the Bush River provided a more conservative estimate of the far-field dilution than the first two models which were based on segmented (complete mixing) approaches.

Total near-field dilutions, which correct for the influence of far-field dilutions (Fisher et al., 1979), were calculated for Maryland's acute and chronic aquatic life toxicity criteria under Spring and Fall conditions. The smaller of the Spring or Fall calculated dilution factors for both the acute and chronic toxicity conditions were then used to estimate the near-field acute and chronic receiving water concentrations of all the contaminants found in the Beach Point groundwater. The estimate for each contaminant was determined by using the highest concentration of the material found in the groundwater (see Section 4.2.4) divided by the dilution factor.

4.2.5.3 Mixing Zone Consideration

Mixing zones are areas where an effluent discharge undergoes initial dilution; they may also be extended to cover secondary mixing in the ambient waterbody. A mixing zone is an allocated impact zone where acute and chronic water quality criteria can be exceeded as long as a number of protections are maintained as required by EPA and the State (U.S. EPA, 1991a). The State of Maryland allows a mixing zone as a policy issue, but requires spatial dimensions to limit the areal extent of the mixing zone. Mixing zones may be allowed on a case-by-case basis. The concept of mixing zones is normally applied in the case of wastewater treatment facilities and power plants that discharge to waterways. Arguments will be presented in Section 5.2.3 that the mixing zone concept could be considered for the groundwater discharge in the Bush River.

SECTION 5

RESULTS AND DISCUSSION

5.1 Biomonitoring Evaluation

The Results and Discussion Section is organized as follows. The results and discussion for all of the biomonitoring systems are presented in separate sections for each test system. The endpoints/responses for each biomonitoring toxicity test are summarized in Table 2. The table is organized as Tests Nos. 1, 2, 3, and 4 which reflects the bimonthly test design. The Microtox®, Japanese medaka LC50, Ames, SCE, and Japanese medaka chronic histopathology and morphometric results are presented under the Test No. 1 column for space convenience purposes only; the tests were not bimonthly tests as the heading implies. The raw data, water quality data, and statistical analyses for the biomonitoring test systems as well as the routine water quality analyses of the histopathological study are given in Burton et al. (1994).

5.1.1 Acute Toxicity Tests

5.1.1.1 Microtox®

A summary of the Microtox® 5- and 15-min EC50 (reduction of bioluminescence) results is given in Table 2. The Microtox® 5- and 15-min EC50s for 100% groundwater at pH 4 ranged from 11-29 and 17-45% groundwater by volume, respectively. The 5- and 15-min EC50s for the 100% groundwater adjusted to pH 7 ranged from 17-59 and 18-96% groundwater by volume, respectively. The groundwater was less toxic at pH 7 than at pH 4. Although the 5- and 15-min EC50s were variable over the course of the study, the raw and buffered groundwater both appeared to become less toxic as the study progressed (Fig. 1 and 2). As will be shown below, the same trend was not evident in the other acute and short-term chronic test systems. Less toxicity was observed in the 15-min EC50s relative to the 5-min EC50s.

Toxicity was detected in 10% groundwater by volume one time during the study. The 5-min EC50 was 99% groundwater by volume; no 15-min EC50 was obtained for the same sample. The EC50 was spurious because the 95% fiducial limits ranged from 0.0 to 1 x 10^6% groundwater by volume. No toxicity was detected during the study in the 1% groundwater by volume samples or the APG-EA diluent water.

The toxicity of the groundwater as shown by the Microtox® assay is not surprising when one considers the complex mixture of the contaminants in the groundwater (Section 5.1.6.1; Table 3). For example, the 5-min EC50s for both copper and zinc are less

TABLE 2. SUMMARY OF THE TOXICITY ENDPOINTS/RESPONSES FOR BIOMONITORING TESTS CONDUCTED ON BEACH POINT GROUNDWATER (WELL CC-33B) FROM FEBRUARY 1993 TO DECEMBER 1993

Bioassay	Endpoint	Value[a]			
		Test No. 1	Test No. 2	Test No. 3	Test No. 4
Microtox®:					
100% (pH 4)	5-min EC50[b]	11-29[c]	N/A	N/A	N/A
100% (pH 7)	5-min EC50[b]	17-59[d]	N/A	N/A	N/A
100% (pH 4)	15-min EC50[b]	17-45[e]	N/A	N/A	N/A
100% (pH 7)	15-min EC50[b]	18-96[f]	N/A	N/A	N/A
10%	5-min EC50[b]	Not Toxic	N/A	N/A	N/A
10%	15-min EC50[b]	Not toxic	N/A	N/A	N/A
1%	5-min EC50[b]	Not toxic	N/A	N/A	N/A
1%	15-min EC50[b]	Not toxic	N/A	N/A	N/A
Diluent H_2O	5-min EC50[b]	Not toxic	N/A	N/A	N/A
Diluent H_2O	15-min EC50[b]	Not toxic	N/A	N/A	N/A
Rotifer:					
pH 4	24-h LC50	72 (64.3-83.4)	64 (57.4-73.5)	70 (62.4-82.0)	95 (75.8-147.6)
pH 7	24-h LC50	Not toxic	No LC50	No LC50	Not toxic
Green alga:					
pH 4	96-h EC50[g]	59 (57.0-61.9)	51 (45.2-58.2)	56 (54.0-59.0)	55 (51.2-58.5)
pH 4	NOEC[g]	56	18	18	18
pH 4	LOEC[g]	100	32	32	32
pH 7	96-h EC50[g]	No EC50	58 (55.8-60.7)	55 (51.1-58.2)	50 (46.6-53.2)
pH 7	NOEC[g]	56	32	18	18
pH 7	LOEC[g]	100	56	32	32

TABLE 2. (CONTINUED)

Bioassay	Endpoint	Value[a] Test No. 1	Test No. 2	Test No. 3	Test No. 4
Cladoceran:					
pH 4	48-h LC50	65 (57.2-75.5)	65 (57.2-75.5)	65 (57.2-75.5)	63 (55.0-73.7)
pH 4	7-d LC50	65 (57.2-75.5)	65 (57.2-75.5)	65 (57.2-75.5)	42 (33.5-53.5)
pH 4	NOEC[h]	18	10	10	10
pH 4	LOEC[h]	32	18	18	18
pH 7	48-h LC50	Not toxic	Not toxic	No LC50	No LC50
pH 7	7-d LC50	43 (1.0-56.4)	72 (53.8-155.8)	No LC50	40 (29.7-52.0)
pH 7	NOEC[h]	[i]	[j]	10	10
pH 7	LOEC[h]	[i]	[j]	18	18
Fathead minnow:					
pH 4	96-h LC50	64 (60.3-68.1)	22 (19.3-24.0)	45 (40.6-49.5)	62 (58.1-65.9)
pH 4	7-d LC50	62 (58.1-65.9)	22 (19.3-24.0)	39 (34.2-43.6)	51 (46.3-57.6)
pH 4	NOEC[k]	32	10	32	32
pH 4	LOEC[k]	56	18	56	56
pH 7	96-h LC50	Not toxic	Not toxic	Not toxic	Not toxic
pH 7	7-d LC50	Not toxic	No LC50	Not toxic	Not toxic
pH 7	NOEC[k]	56	56	Not toxic	Not toxic
pH 7	LOEC[k]	100	100	Not toxic	Not toxic
Japanese medaka:					
pH 4	96-h LC50	No LC50	N/A	N/A	N/A

TABLE 2. (CONTINUED)

Bioassay	Endpoint	Value[a]			
		Test No. 1	Test No. 2	Test No. 3	Test No. 4
Ames:					
100%	Mutagenicity	[l]	[l]	Negative	Negative
100% (10X)	Mutagenicity	[l]	[l]	Negative	Negative
10%	Mutagenicity	Negative	Negative	Negative	Negative
10% (10X)	Mutagenicity	Negative	Negative	Negative	Negative
1%	Mutagenicity	Negative	Negative	[l]	[l]
1% (10X)	Mutagenicity	Negative	Negative	[l]	[l]
Diluent H_2O	Mutagenicity	Negative	Negative	Negative	Negative
Diluent H_2O (10X)	Mutagenicity	Negative	Negative	Negative	Negative
SCE:					
100% (Conc)	DNA damage	Positive	N/A	N/A	N/A
100%	DNA damage	Negative	N/A	N/A	N/A
10% (Conc)	DNA damage	Negative	N/A	N/A	N/A
Diluent H_2O (Conc)	DNA damage	Negative	N/A	N/A	N/A
FETAX:					
pH 4	4-d LC50	Not toxic	Not toxic	Not toxic	No LC50
pH 4	4-d EC50[m]	No EC50	89 (71.1-143.1)	No EC50	No EC50
pH 4	NOEC[n]	10	10	10	32
pH 4	LOEC[n]	18	18	18	56
pH 7	4-d LC50	Not toxic	No LC50	Not toxic	No LC50
pH 7	4-d EC50[m]	No EC50	No EC50	No EC50	No EC50
pH 7	NOEC[n]	18	18	18	18
pH 7	LOEC[n]	32	32	32	32

TABLE 2. (CONTINUED)

Bioassay	Endpoint	Value[a]			
		Test No. 1	Test No. 2	Test No. 3	Test No. 4
Chronic histopathology and growth:					
10%	6/9-m Wet weight	No effect	N/A	N/A	N/A
1%	6/9-m Wet weight	No effect	N/A	N/A	N/A
Diluent H_2O	6/9-m Wet weight	N/A	N/A	N/A	N/A
10%	6/9-m Std. lgth.	No effect	N/A	N/A	N/A
1%	6/9-m Std. lgth.	No effect	N/A	N/A	N/A
Diluent H_2O	6/9-m Std. lgth.	N/A	N/A	N/A	N/A
10%	6/9-m Lesions	Negative[o]	N/A	N/A	N/A
1%	6/9-m Lesions	Negative[o]	N/A	N/A	N/A
Diluent H_2O	6/9-m Lesions	Negative[o]	N/A	N/A	N/A

[a] All endpoints are given as percent groundwater by volume. With the exception of the Microtox® 95% fiducial limits which are given in the footnotes, the 95% confidence limits for each LC50 and EC50 value are presented directly below the value in parentheses.

[b] Range of all EC50s for reduction in bioluminescent activity conducted from April-December.

[c] The 95% fiducial limits for the 5-min EC50s of 11 and 29 at pH 4 are 7.6-14.8 and 21.6-39.2, respectively.

[d] The 95% fiducial limits for the 5-min EC50s of 17 and 59 at pH 7 are 7.9-36.2 and 46.3-75.4, respectively.

[e] The 95% fiducial limits for the 15-min EC50s of 17 and 45 at pH 4 are 7.4-39.0 and 12.0-169.6, respectively.

[f] The 95% fiducial limits for the 15-min EC50s of 18 and 96 t pH 7 are 9.6-35.1 and 30.1-303.8., respectively.

[g] Test endpoint- reduction in growth (cell density).

[h] Test endpoint- reduction in neonate production.

TABLE 2. (CONTINUED)

Bioassay	Endpoint	Value[a]			
		Test No. 1	Test No. 2	Test No. 3	Test No. 4

[i] The NOEC and LOEC could not be obtained because significant (α = 0.05) reductions in neonate production relative to the controls occurred at all concentrations down to 32% buffered groundwater by volume, which was the lowest concentration studied.

[j] The NOEC and LOEC could not be obtained because significant (α = 0.05) reductions in neonate production relative to the controls occurred at all concentrations down to 18% buffered groundwater by volume, which was the lowest concentration studied.

[k] Test endpoint- reduction in growth (dry weight) for Test Nos. 1 and 2 at pH 4 and Test No. 1 at pH 7; the test endpoint for Test Nos. 3 and 4 at pH 4 and Test No. 2 at pH 7 was an increase in mortality rather than a reduction in growth.

[l] Assay not conducted.

[m] 96-h EC50 for malformations.

[n] Test endpoint- increased number of malformations.

[o] See Section 5.1.5 for additional information.

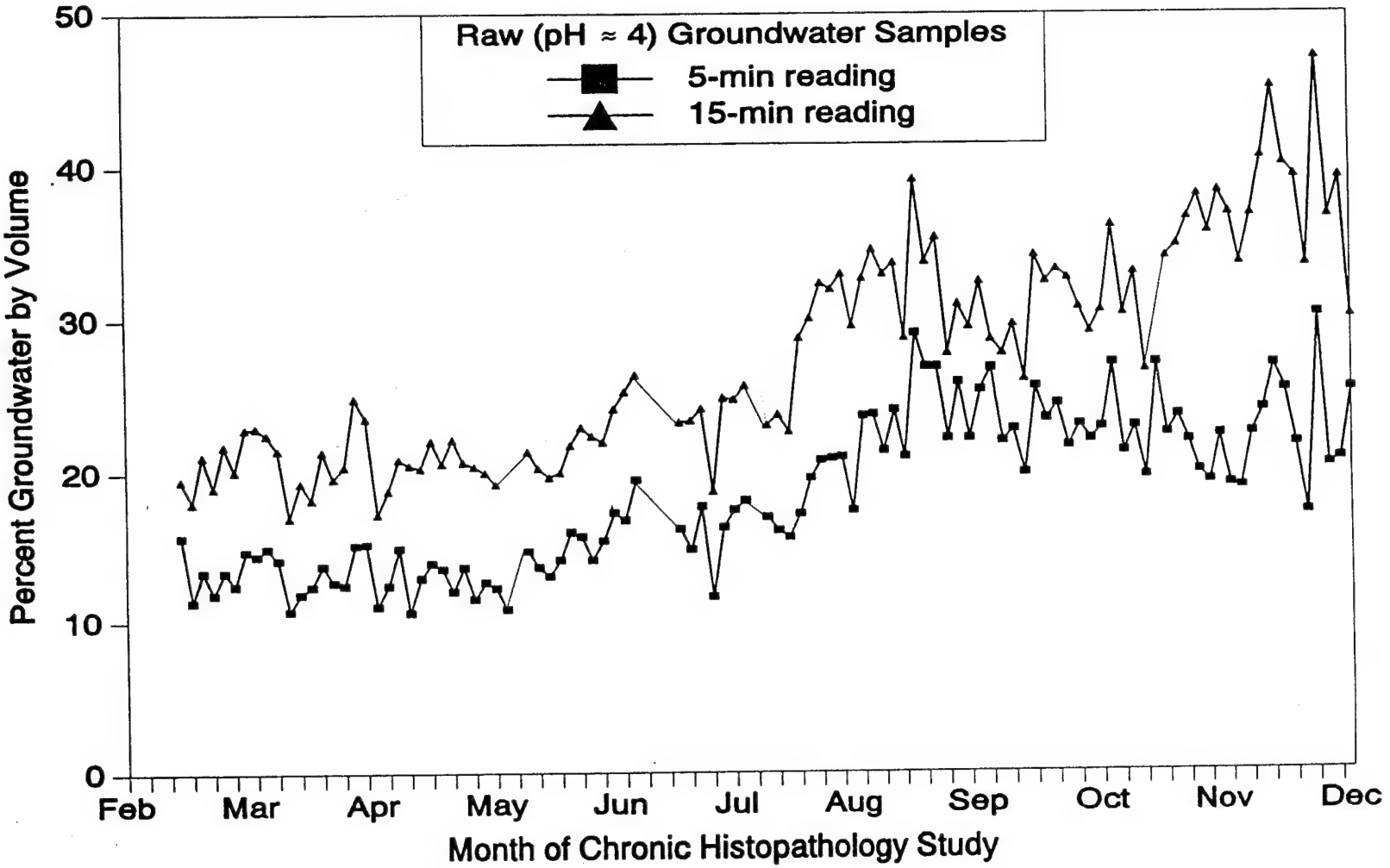

Figure 1. Microtox® test results of raw (pH ≈ 4) groundwater samples taken during the chronic histopathology study.

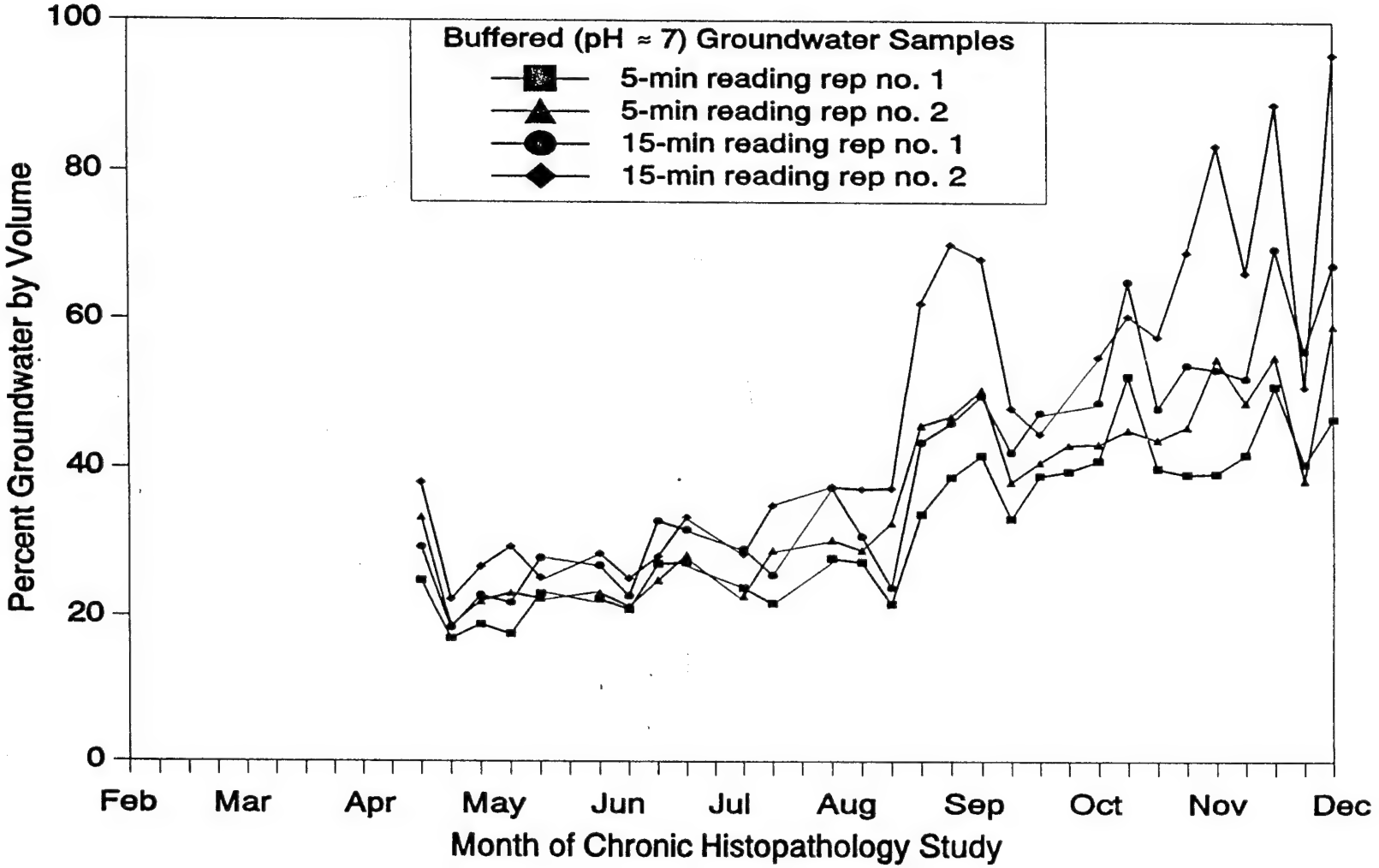

Figure 2. Microtox® test results of buffered (pH ≈ 7) groundwater samples taken during the chronic histopathology study.

than the groundwater concentrations shown in Table 3 (Qureshi et al., 1982). Similarly, several of the volatile organics at concentrations below the concentrations in the raw groundwater have been shown to be toxic via the 5- and/or 15-min Microtox® analysis (Kaiser and Ribo, 1988). Chloroform, chlorobenzene, trichloroethane, and tetrachloroethane all have 5- and/or 15-min EC50s below the concentrations found in the groundwater.

5.1.1.2 Rotifer

The results of the 24-h LC50 tests at pH 4 and 7 with the rotifer are summarized in Table 2. The 24-h LC50 values (95% confidence limits given in parenthesis) for the four acute rotifer tests conducted at pH 4 ranged from 64-95% groundwater by volume. No toxicity was found at pH 7 in the first and fourth bimonthly tests. Some mortality occurred in the 100% groundwater treatment during the second and third tests; however, no LC50s could be calculated because <50% mortality occurred.

5.1.1.3 Green Alga, Cladoceran, Fathead Minnow, and Japanese Medaka

The 96-h EC50s (reduction in growth) for the green alga exposed to raw groundwater at pH 4 and buffered groundwater at pH 7 are given in Table 2. The 96-h EC50s for reduction in growth at pH 4 ranged from 51-59% groundwater by volume. With the exception of Test No. 1 in which no EC50 could be calculated because <50% reduction in growth occurred in the study, the 96-h EC50s were essentially the same in the groundwater buffered to pH 7. The EC50s at pH 7 in Test Nos. 2-4 ranged from 50-58% groundwater by volume.

Groundwater at pH 4 was acutely toxic to the cladoceran (Table 2). The 48-h LC50s ranged from 63-65% groundwater by volume for the four tests. Buffered groundwater was not toxic in Test Nos. 1 and 2. Some toxicity occurred at pH 7 in Tests Nos. 3 and 4; however, no 48-h LC50 could be calculated because <50% mortality occurred in 48 h.

The 96-h LC50 data for the fathead minnow at pH 4 and 7 are summarized in Table 2. Groundwater at pH 4 was acutely toxic to the fathead minnow. The 96-h LC50s ranged from 22-64% groundwater by volume for the four tests. Buffered groundwater at pH 7 was not toxic in any test after a 96-h exposure.

One 96-h LC50 toxicity test was conducted with Japanese medaka at pH 4 (Table 2). The groundwater was toxic at pH 4 after a 96-h exposure; however, a 96-h LC50 could not be calculated because <50% mortality occurred in 96 h. An acute toxicity test was not conducted at pH 7 since the groundwater was not acutely toxic to Japanese medaka at pH 4.

TABLE 3. SUMMARY OF THE FOUR BIMONTHLY CHEMICAL ANALYSES (RANGE OF CONCENTRATIONS) CONDUCTED ON RAW BEACH POINT GROUNDWATER (WELL CC-33B) FROM FEBRUARY 1993 TO DECEMBER 1993

Parameter	Concentration	Unit
	General Water Quality	
Alkalinity	<1 - 21	mg/L as $CaCO_3$
Hardness	170 - 230	mg/L as $CaCO_3$
Conductivity	1,600 - 2,020	µs/cm
Total Suspended Solids	<5	mg/L
Total Organic Carbon	2 - 4	mg/L
Calcium	16.5 - 21	mg/L as Ca
Magnesium	30 - 43	mg/L as Mg
Manganese	1.7 - 2.1	mg/L as Mn
Potassium	2.5 - 4.1	mg/L as K
Sodium	257 - 350	mg/L as Na
Sulfate	62 - 91	mg/L as SO_4
Sulfide	<1	mg/L as S
Chloride	490 - 580	mg/L as CL
Ammonia-Nitrogen	<0.1	mg/L as N
Nitrate-Nitrogen	0.1 - 2.5	mg/L as N
Nitrite-Nitrogen	<0.01	mg/L as N
Phosphorus	0.12 - 2.64	mg/L as P
Fluoride	0.2 - 1.2	mg/L as F
Iron	4.6 - 7.7	mg/L as Fe
Cyanide	<0.01 - 0.06	mg/L as CN
Bromide	0.4 - 1.5	mg/L as Br
pH	3.9 - 4.1	Std. Unit

TABLE 3. (CONTINUED)

Parameter	Concentration	Unit
	Heavy Metals[a]	
Aluminum	810 - 1,500	µg/L as Al
Antimony	<5	µg/L as Sb
Arsenic	<5-7	µg/L as As
Beryllium	<5	µg/L as Be
Boron	50 - 70	µg/L as B
Cadmium	0.8 - 1.4	µg/L as Cd
Chromium	<2	µg/L as Cr
Cobalt	80 - 110	µg/L as Co
Copper	5.4 - 8	µg/L as Cu
Lead	<5	µg/L as Pb
Mercury	<0.5	µg/L as Hg
Molybdenum	<10	µg/L as Mo
Nickel	120 - 180	µg/L as Ni
Selenium	<5	µg/L as Se
Silver	<1	µg/L as Ag
Thallium	<5	µg/L as Ti
Tin	<50	µg/L as Sn
Zinc	210 - 310	µg/L as Zn
	Volatile Organics	
Chlorobenzene	2	µg/L
Chloroform	10 - 12	µg/L
1,1,2-Trichloroethane	37 - 110	µg/L
1,1,1,2-Tetrachloroethane	3	µg/L
1,1,2,2-Tetrachloroethane	9,000 - 17,000	µg/L

TABLE 3. (CONTINUED)

Parameter	Concentration	Unit
	Volatile Organics Con't	
1,1-Dichloroethene	1	µg/L
cis-1,2-Dichloroethene	110 -640	µg/L
Trichloroethene	450 - 1,600	µg/L
Tetrachloroethene	30 - 90	µg/L
Vinyl Chloride	2	µg/L

[a] The metal concentrations are total metal; not dissolved metal.

The acute toxicity of the groundwater to the rotifer, cladoceran, and fathead minnow at pH 4 may be attributable to the heavy metals in the groundwater (Table 3). Several EPA priority pollutant heavy metals (cadmium, copper, nickel, and zinc) were found in the groundwater. The concentration of zinc in the groundwater (when adjusted for hardness) exceeded the EPA numerical water quality criterion of 120 µg/L for freshwater invertebrates and fish (U.S. EPA, 1987). Furthermore, metals such as copper and zinc exist primarily as divalent cations at a pH of 4 which is the most toxic form of the metal (Lee, 1973; Sprague, 1985). It is well established that the toxicity of metals in chemical mixtures is additive for many aquatic animals (Marking, 1985). It is likely that the toxicity observed in the study may have been additive or greater than additive (de March, 1988). The reduction and/or elimination of toxicity at pH 7 for the same test organisms is most likely related to the reduction in toxicity of heavy metals as they shift from a divalent cation at pH 4 to less toxic species at pH 7 (Lee, 1973). In contrast to the acute toxicity of the groundwater at pH 4 to the rotifer, cladoceran, and fathead minnow, a 96-h LC50 could not be calculated for the Japanese medaka because <50% mortality occurred.

No single heavy metal at the concentrations present in the groundwater can account for the toxicity observed in the green alga (U.S. EPA, 1986b). Few data are available on the possible joint toxicity of heavy metals or organics to green algae (Faust et al., 1994). The toxicity at pH 4 and 7 was essentially the same (Table 2) which suggests that heavy metals may not be

important toxicologically to the alga. No single priority pollutant organic for which there are toxicity data can account for the toxicity observed in the alga (U.S. EPA, 1986b). One can only speculate that the toxicity of the groundwater to the alga may be related to 1) the joint toxicity of the metals, 2) joint toxicity of the organics and/or 3) interaction of the metals and organics.

5.1.2 Short-term Chronic Toxicity Tests

5.1.2.1 Green Alga

The NOECs and LOECs (reduction in cell density) for the green alga exposed to groundwater at pH 4 and 7 are summarized in Table 2. At pH 4, the NOECs for the four tests ranged from 18-56% groundwater by volume. The LOECs ranged from 32-100% groundwater by volume. At pH 7, the NOECs and LOECs, respectively, ranged from 18-56 and 32-100% groundwater by volume. With the exception of Test No. 2, the NOECs and LOECs were exactly the same at pH 4 and 7 for the same test period. The NOECs and LOECs were less at pH 7 than at pH 4 in Test No. 2 only. Thus, with the exception of one test, no difference in algal toxicity was found for groundwater at pH 4 and pH 7. Some variation in toxicity was found between the four sets of tests. The lowest NOEC found in any test was 18% groundwater by volume. As stated above in Section 5.1.1.3, one can only speculate that the toxicity of the groundwater to the alga may be related to the 1) joint toxicity of the metals, 2) joint toxicity of the organics and/or 3) interaction of the metals and organics.

5.1.2.2 Cladoceran

The 7-d LC50s, NOECs, and LOECs for the cladoceran exposed to groundwater at pH 4 and 7 are summarized in Table 2. The groundwater at pH 4 was toxic in all tests. The short-term chronic 7-d LC50s at pH 4 ranged from 42-65% groundwater by volume. The NOECs (reduction in neonate production) at pH 4 ranged from 10-32% groundwater by volume while the LOECs ranged from 18-32% groundwater by volume in the four tests.

The groundwater at pH 7 was essentially as toxic to the cladoceran as it was at pH 4. The 7-d LC50s ranged from 40-72% groundwater by volume in three of the four tests. Toxicity was observed in Test No. 3; however, the 7-d LC50 could not be calculated because <50% mortality occurred after 7 d of exposure. The NOECs and LOECs in Test Nos. 1 and 2 could not be determined because significant reductions ($\alpha = 0.05$) in neonate production occurred down to 32% and 18% groundwater by volume, respectively, which were the lowest concentrations used in the study. The NOECs and LOECs in Test Nos. 3 and 4 were both 10% and 18%, respectively. The NOECs and LOECs were the same in Test Nos. 3 and 4 in both raw and buffered groundwater.

5.1.2.3 Fathead Minnow

The 7-d LC50s, NOECs, and LOECs for the fathead minnow exposed to groundwater at pH 4 and 7 are summarized in Table 2. The groundwater at pH 4 was toxic in all tests. The short-term chronic 7-d LC50s at pH 4 ranged from 22-62% groundwater by volume. The NOECs at pH 4 ranged from 10-32% groundwater by volume while the LOECs ranged from 18-56% groundwater by volume in the four tests. The NOEC and LOEC endpoints at pH 4 were a reduction in growth for Test Nos. 1 and 2 and an increase in mortality in Test Nos. 3 and 4.

The groundwater was less toxic when buffered to pH 7 (Table 2). With regard to the 7-d LC50s at pH 7, no statistically significant ($\alpha = 0.05$) mortality occurred in three of the four tests. Some mortality was observed in Test No. 2; however, a 7-d LC50 could not be calculated because <50% mortality occurred in 7 d of exposure. A significant reduction ($\alpha = 0.05$) in growth occurred in Test No. 1 while a significant increase in mortality occurred in Test No. 2. The NOECs for Test Nos. 1 and 2 were 56% groundwater by volume, while the LOECs were 100% groundwater by volume. No NOECs or LOECs were obtained in the third and fourth tests because statistically significant ($\alpha = 0.05$) increases in mortality or reductions in growth did not occur.

The chronic toxicity observed for the cladoceran and fathead minnow may be related to several heavy metals and at least one organic present in the groundwater (Table 3). As discussed below in Section 5.1.6.1, cadmium, nickel, and zinc concentrations in the groundwater are equal to or exceed the EPA freshwater chronic numerical water quality criteria of 1.1, 160, and 110 µg/L (hardness dependent criteria; 100 mg/L as $CaCO_3$ used), respectively (U.S. EPA, 1980a; 1984a; 1987). None of the priority pollutant organics found in the groundwater currently have numerical water quality criteria values because insufficient data exist to develop criteria (Potts, 1994). However, 1,1,2,2-tetrachlorethane concentrations found in the groundwater exceed the EPA freshwater chronic LOEC (U.S. EPA, 1980b). Thus, it likely that the chronic toxicity observed in the cladoceran and fathead minnow tests is a combination of metals and organics.

In contrast to the general reduction in acute toxicity when the rotifer and fathead minnow were tested in buffered groundwater, the chronic toxicity NOECs and LOECs values for the cladoceran were the same at pH 4 and 7 (Table 2). Thus, the suggestion above that toxicity attributable to heavy metals is reduced at the higher pH does not appear to be valid for the cladoceran in the chronic tests. The reason for this observation is not clear.

5.1.3 Genotoxicity Tests

5.1.3.1 Gene Mutation Assay

Both unconcentrated and concentrated (10X) groundwater Ames assays were conducted on 10% groundwater by volume, 1% groundwater by volume, and control water in Test Nos. 1 and 2; 100% groundwater, 10% groundwater by volume, and control water were assayed in Test Nos. 3 and 4 (Table 2). All of the unconcentrated and concentrated (10X) 100% groundwater, 10% groundwater by volume, 1% groundwater by volume, and diluent water assays were found to be non-mutagenic (negative) with tester strains TA98 and TA100 in both the presence and absence of an exogenous metabolic activation system of mammalian microsomal enzymes derived from Aroclor-induced rat liver (S9 mix). One deviation from the above results was found in the unconcentrated APG-EA diluent water in Test No. 1. With tester strain TA98 in the absence of S9 mix, the response was evaluated as marginal in two trials due to the presence of 1 dose with an MR $\geq$ 1.5 which was not accompanied by a dose response.

5.1.3.2 Sister Chromatid Exchange Assay

The sister chromatid exchange (SCE) assay was preformed once on concentrated samples ($\approx$50,000X) of 100% groundwater, 10% groundwater by volume, APG-EA diluent water, and once on an unconcentrated 100% groundwater sample. The concentrated extract of 100% groundwater was considered positive for inducing sister chromatid exchange in Chinese hamster ovary cells under the nonactivation conditions of the study and weakly positive under the activation (exogenous metabolic activation system of mammalian microsomal enzymes derived from Aroclor-induced rat liver S9 mix) conditions of the study (Table 2). The unconcentrated sample of 100% groundwater was considered negative for inducing sister chromatid exchange in Chinese hamster ovary cells under the activation and nonactivation conditions of the assay. The concentrated extracts of the 10% groundwater by volume and APG-EA diluent water were also considered negative for inducing sister chromatid exchange in Chinese hamster ovary cells under the activation and nonactivation conditions of the assay.

A positive response was found for 100% groundwater in the SCE assay when the groundwater sample was concentrated $\approx$50,000X. The SCE assay was negative for unconcentrated 100% groundwater. It is not clear that the SCE positive response in the 50,000X concentrated sample is important to aquatic organisms. As discussed below in Section 5.2, the potential for the heavy metals and aliphatic organics present in the groundwater to bioaccumulate is <100-fold because the log K_{ow}s of the materials are $\leq$3. It is clear that no chemical or physical process in the Bush River could cause a 50,000X increase in the concentration of the materials in the near-field of the groundwater discharge.

Thus, organisms in the immediate vicinity of the groundwater discharge would never be exposed to a 50,000X increase in the contaminants.

5.1.4 Developmental Toxicity Test

The 4-d LC50, 4-d EC50 (malformations), NOEC, and LOEC results for the FETAX assays conducted in groundwater at pH 4 and 7 are summarized in Table 2. Little embryolethality occurred in the groundwater at pH 4 or 7 (Table 2). The raw groundwater was not toxic to the embryos at pH 4 in the first three tests. Some toxicity occurred in Test No. 4; however, a LC50 could not be calculated because <50% mortality occurred. The buffered groundwater was not toxic in two of the four test. Some mortality occurred in Test Nos. 2 and 4; however, a LC50 could not be calculated.

Significant ($\alpha = 0.05$) embryo malformations occurred in the raw and buffered groundwater in all assays. A 96-h EC50 (malformations) of 89% groundwater by volume was obtained in the second test at pH 4; 96-h EC50s could not be calculated for any of the other tests. The NOECs and LOECs for the four groundwater tests at pH 4 were 10 and 18% groundwater by volume, respectively, in the first three tests and 32 and 56% groundwater by volume in Test No. 4. The NOECs and LOECs for the four groundwater tests at pH 7 were all 18 and 32% groundwater by volume, respectively. With the exception of the fourth test, the buffered groundwater was less toxic than the raw groundwater. For some unexplainable reason, the raw groundwater was less toxic than the buffered groundwater in Test No. 4.

The types of malformed embryos (as described by Bantle et al., 1991) after 96 h of exposure in raw groundwater were primarily coiled guts, multiple edema, and abdominal edema. Severe, notochordal and facial malformations were also observed in <10% of the total malformations. Fewer malformations were observed in the buffered groundwater tests; however, the same types of malformations that occurred in the raw groundwater were observed in the buffered groundwater assays (primarily coiled guts, multiple and abdominal edema with <10% severe, notochord, and facial). The incidences of malformations were greater at the higher test concentrations in both the raw and buffered groundwater.

The developmental toxicity found in the FETAX assays is most likely related to the heavy metals present in the groundwater. Several heavy metals, including copper, cadmium, and zinc, have been shown to cause developmental problems in lower vertebrate aquatic organisms (Weis and Weis, 1989). Dawson et al. (1985) found that mixtures of heavy metals (copper, cadmium, lead, and zinc) from acidic mine sources caused teratogenic effects and mortality when evaluated by FETAX. When the pH was adjusted from

lows which ranged from 3.2 to 5.9 to pH 7, toxicity and teratogenicity decreased. The same response occurred in the present study with the exception of Test No. 4 in which the raw groundwater was less toxic than the buffered groundwater. The possible role of the organics in the groundwater is not clear since FETAX data do not exist for the individual materials (Bantle, 1994).

5.1.5 Chronic Histopathology and Growth Test

The mean wet weight of the Japanese medaka after 6 months of exposure to 10% groundwater by volume, 1% groundwater by volume, and diluent water was 487, 484, and 477 mg, respectively. Mean standard lengths after 6 months of exposure were, 31, 30, and 31 mm, respectively, for fish exposed to 10% groundwater by volume, 1% groundwater by volume, and diluent water. Mean wet weights after 9 months of exposure to 10% groundwater by volume, 1% groundwater by volume, and diluent water were 584, 562, and 573 mg. Mean standard lengths after 9 months of exposure were 33, 32, and 32 mm, respectively, for fish exposed to 10% groundwater by volume, 1% groundwater by volume, and diluent water.

No significant difference ($\alpha = 0.05$) in wet weight or standard length was found between treatments after 6 and 9 months of exposure (Table 2). With regard to the 9-month length data only, a statistically significant difference ($\alpha = 0.003$) in standard length was found using the nonparametric Kruskal Wallis test. However, when Dunn's multiple comparison test was used to enumerate the difference between the control and each experimental treatment, no statistically significant difference ($\alpha = 0.05$) was found. Standard length was judged not to be affected by a 9-month exposure to the groundwater because 1) Dunn's test could not discern any statistical difference ($\alpha = 0.05$) between groups and 2) the numerical differences between mean lengths in the study groups were quite small.

The cumulative mortality of Japanese medaka during the 9-month study is shown graphically in Figure 3. Percent cumulative mortality is given in Figure 4. Cumulative percent mortality at 6 months for the controls, fish exposed to 1% groundwater by volume, and 10% groundwater by volume was 4.5, 2.5, and 2.1%, respectively. Cumulative percent mortality at 9 months for the controls, fish exposed to 1% groundwater by volume, and 10% groundwater by volume was 22.1, 11.7, and 7.2%, respectively. To the authors knowledge, there are no test mortality acceptability criteria for a 9-month test. If one uses the draft ASTM standard guide mortality acceptability criteria for early life stage (ELS) toxicity tests which run for 1-2 months after hatch or fry swim-up, the mortality observed in this study falls within ELS acceptability criteria (Goodman, 1986). For example, the ELS test acceptability criteria for all eight freshwater species listed in the draft ASTM standard guide (Japanese medaka are not

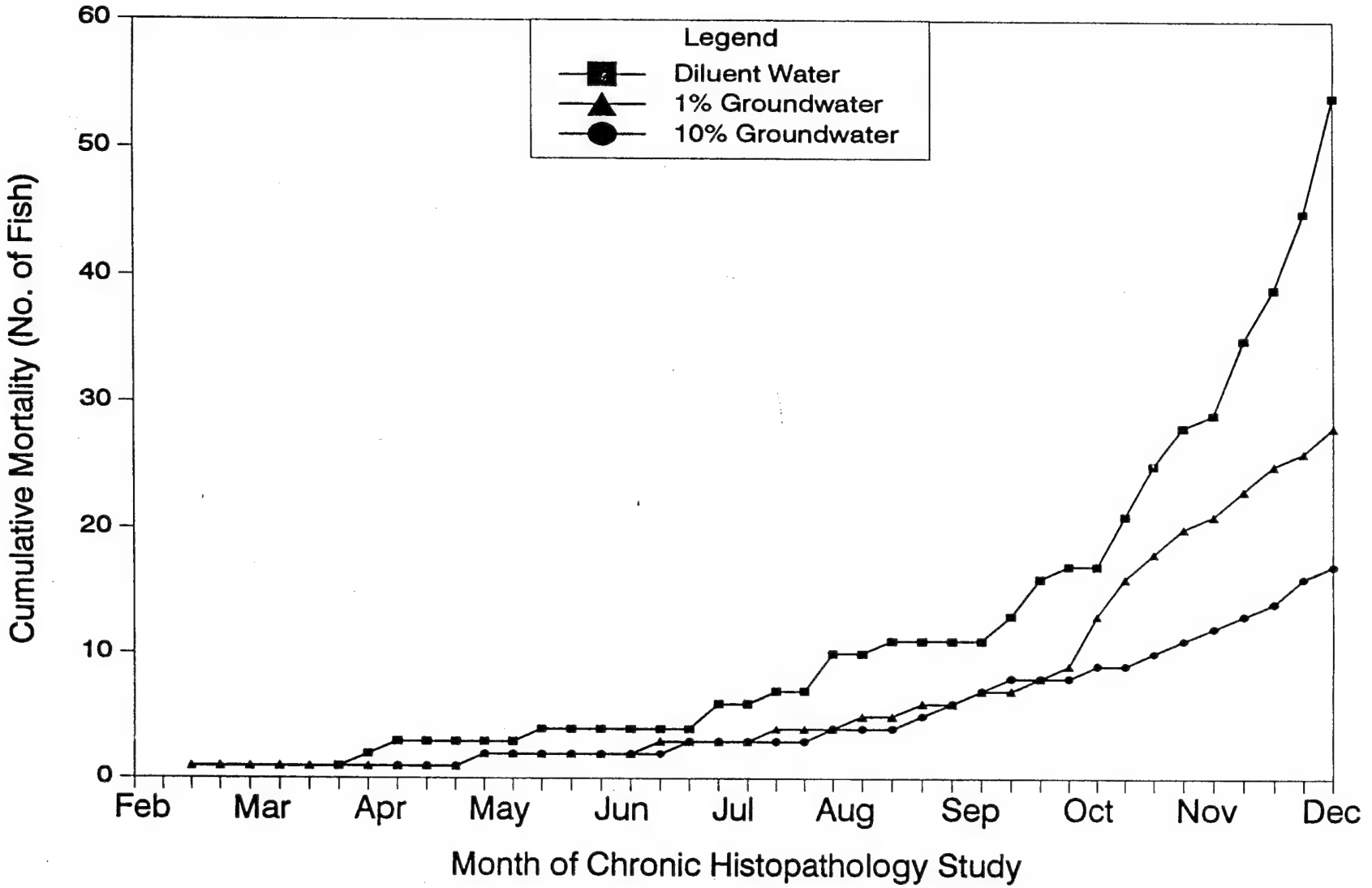

Figure 3. Cumulative mortality of Japanese medaka during the 9-month chronic histopathology exposure study.

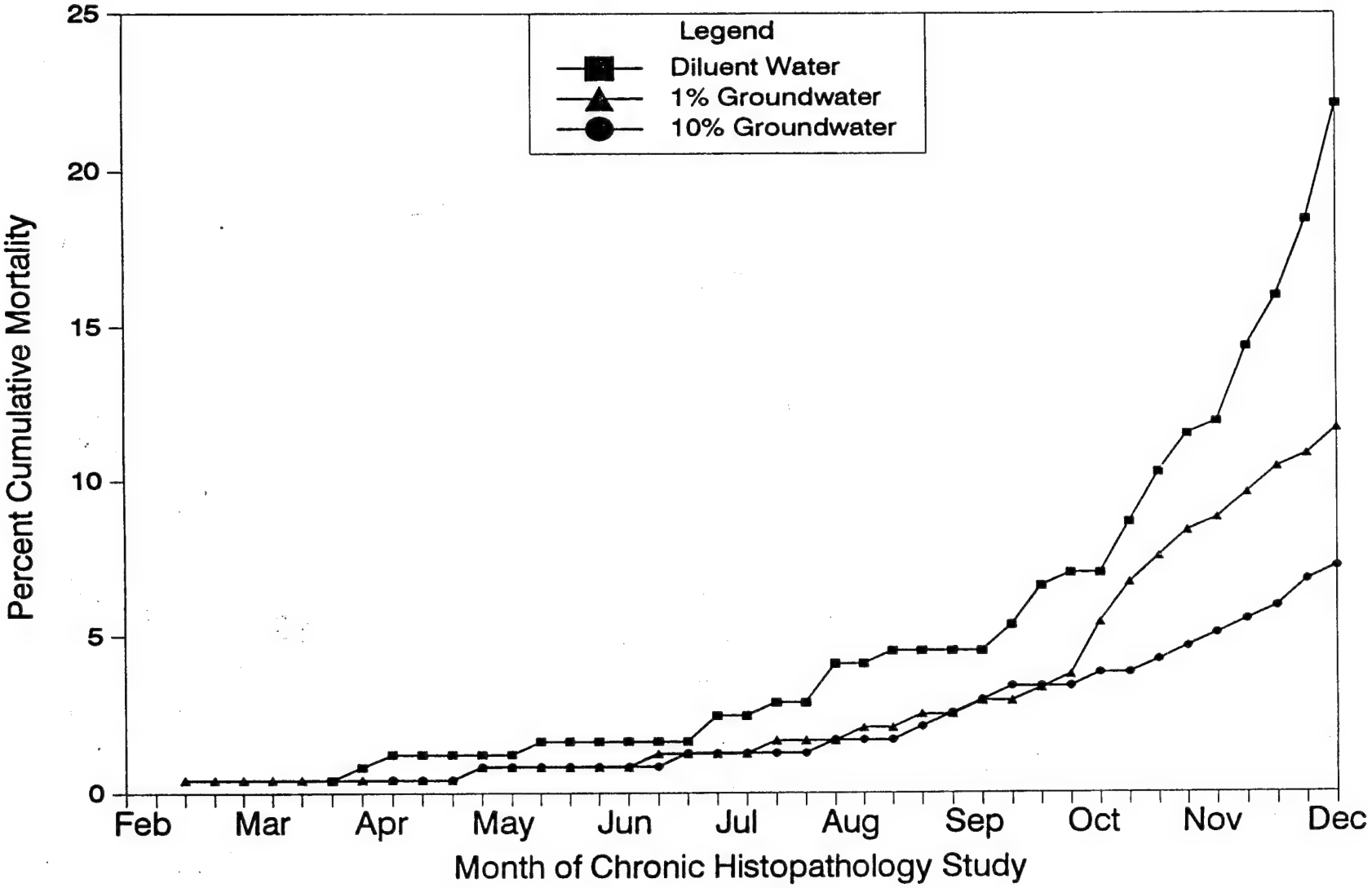

Figure 4. Percent cumulative mortality of Japanese medaka during the 9-month chronic histopathology exposure study.

included in the guide) run from 60-75% (Goodman, 1986). That is, for a test to be acceptable, 60-75% of the control organisms must be alive at the end of the study depending on the species being studied. Excluding fish sacrificed at 6 months, 77.9% of the control Japanese medaka were living at the end of the 9-month period.

The histopathological findings for the 6- and 9-month exposures are described in detail in the pathology report by Experimental Pathology Laboratories, Inc. (1994). Briefly, the major conclusions of the histopathology study are as follows. After 6 months of exposure to control water, 1% groundwater by volume, and 10% groundwater by volume, liver neoplasia (hepatocellur adenoma and carcinoma) occurred only among Japanese medaka that had been previously initiated with DEN. Seven of 40 medaka were affected in the fish exposed to 10% groundwater by volume as compared to 2 of 40 among the controls and 1 of 40 Japanese medaka exposed to 1% groundwater by volume.

After 9 months of exposure, hepatocellular adenomas or carcinomas, single or multiple, occurred in DEN-initiated Japanese medaka that had been exposed to control water, 1% groundwater by volume, and 10% groundwater by volume. There were as many neoplasms among the controls (8 of 122) as among the 10% exposure group (8 of 144). There were two liver neoplasms (1 hepatocellular adenoma and 1 carcinoma) in Japanese medaka exposed to 10% groundwater by volume (no DEN exposure). Two additional liver neoplasms not related to hepatocytes occurred in the DEN 10% groundwater by volume group. One was a cholangioma and the other was a carcinoma, cell of origin unknown. In the DEN 1% groundwater by volume exposure groups there were five (5 of 144) hepatocellular neoplasms, all adenomas. At 9 months there was no dose-related response for liver neoplasia in Japanese medaka exposed to groundwater.

Other neoplasms that occurred sporadically among the various exposure groups at 6 and 9 months included lymphosarcoma, seminoma, thyroid follicular cell adenoma, thyroid follicular cell carcinoma, hemangioma of the gills, and a gallbladder papilloma.

Granulomatous inflammation of the subcutaneous tissue of the lower and/or upper jaw occurred in small numbers of male fish at 6 months. The same lesion occurred among Japanese medaka of all exposure groups at 9 months. The incidence was substantially greater among males than among females. For example, 144 males overall versus 17 females were diagnosed with granulomatous inflammation of the lower jaw. Granulomatous inflammation or granulomas occurred in a variety of additional tissues among all exposure groups. The reason for the higher incidence of these lesions in males versus females is not known. Trauma may be an initiating factor in the granulomatous lesions of the jaws, but

the reason why the jaws of these fish should be traumatized is not known.

Granulomas in the heart, hematopoietic tissue, liver, and spleen occurred in the highest incidence in the controls and in the lowest incidence in Japanese medaka exposed to 10% groundwater by volume. The reasons for this lower incidence among fish exposed to 10% groundwater by volume as compared to the controls is not known.

The DEN-induced hepatic neoplasia found in this study have been observed by several investigators using the Japanese medaka as a model to evaluate carcinogenic potential of selected chemicals in aquatic organisms (for ex., Bunton, 1990; Lauren et al., 1990; Hinton et al., 1988). Protocols for studying DEN hepatocarcinogenicity vary but generally involve exposure to 10-100 mg/L DEN in ambient water for several days or weeks followed by several weeks for tumor development (Hinton et al., 1985). The DEN exposure period in this study was 48 h, which is substantially shorter than the exposure periods normally used by other investigators. Ichikawa and Takayama (1979) have shown that the length of exposure and length of grow-out can enhance hepatic neoplasia induced by DEN.

Little, if any, data are available on the potential carcinogenicity of heavy metals to the Japanese medaka (Hawkins, 1994). Similarly, short-chain halogenated hydrocarbons have received little study relative to other organic groups, e.g., nitroso compounds, polynuclear aromatic hydrocarbons, and aromatic amines (Walker et al., 1985). 1,1,2,2-Tetrachloroethane was not found to be carcinogenic to Japanese medaka exposed to concentrations up to 14 mg/L in a recent study by Hawkins (1991).

5.1.6 Chemical Analyses

5.1.6.1 Comprehensive Chemical Analyses

A summary of the raw groundwater general water quality, heavy metals, and volatile organics measured in the samples of the four bimonthly comprehensive chemical analyses is given in Table 3. The range of the lowest and highest concentrations of the four analyses is presented. The comprehensive results of each bimonthly chemical analysis, including the values for 100% groundwater, 10% groundwater by volume, 1% groundwater by volume, and diluent water are given in Appendix 1, Tables A1-1, A1-2, A1-3, and A1-4 for Test Nos. 1, 2, 3, and 4, respectively. The tables in Appendix 1 include the test method and detection limit for each chemical. In contrast to Table 3, which contains only the range of materials actually measured in the samples, all materials measured and quantified as well as materials not detected during analysis are included in Appendix 1.

No compounds in the following groups were detected at EPA's quantitation limits for groundwater: 1) acid or base/neutral compounds; 2) pesticides; 3) herbicides; or 4) organo-phosphorus pesticides. The following munitions were not detected at a quantitation limit of 50 µg/L: 1) octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine (HMX); 2) hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX); 3) 1,3,5-trinitrobenzene (TNB); 4) N,2,4,6-tetranitro-N-methylaniline (tetryl); 5) trinitrotoluene (TNT); 6) 2,4-dinitrotoluene (2,4-DNT)); or 7) 2,6-dinitro-toluene (2,6-DNT).

The general water chemistry parameters of the groundwater summarized in Table 3 show that the groundwater is a moderately hard water (hardness = 170-230 mg/L as $CaCO_3$). The pH of 3.9-4.1 is low relative to that which occurs in most surfaces waters. Some surface waters high in tannic acid or those waters impacted by acid rain may also have pH values in the same range (Baker et al., 1990). Ammonia-nitrogen was <0.1 mg/L in all samples; thus, nonionized ammonia would not be expected to play a role in toxicity (Thurston et al., 1979).

Several EPA priority pollutant heavy metals were found in the groundwater (Table 3). The concentrations of cadmium, copper, nickel, and zinc exceeded one or more of EPA's numerical water quality criteria for the specific metal. Cadmium, nickel, and zinc concentrations were equal to or exceeded the EPA freshwater chronic numerical water quality criteria of 1.1, 160, and 110 µg/L (hardness dependent criteria; 100 mg/L as $CaCO_3$ used), respectively (U.S. EPA, 1980a; 1984a; 1987). Zinc also exceeded the freshwater acute criteria of 120 µg/L (hardness dependent criteria; 100 mg/L as $CaCO_3$ used). Copper exceeded EPA's saltwater acute criterion of 2.9 µg/L (U.S. EPA, 1984b); a chronic saltwater copper criterion does not exist (Potts, 1994). Aluminum was also present at high concentrations in the groundwater; however, EPA has not established numerical water quality criteria for the metal in freshwater or saltwater under acute or chronic exposure conditions (Potts, 1994).

A number of chlorinated aliphatic compounds were found in the groundwater (Table 3). Several of the organics were EPA priority pollutants. None of the priority pollutant organics found in the groundwater currently have numerical water quality criteria values because insufficient data exist to develop criteria (Potts, 1994). EPA does give the LOEC for several of the compounds where criteria are not available. 1,1,2,2-Tetrachlorethane concentrations found in the groundwater exceed the EPA freshwater chronic LOEC of 2,400 µg/L; no freshwater acute LOEC is given by EPA (U.S. EPA, 1980b). None of the 10 volatile organics found in the groundwater have octanol water partition coefficients (log k_{ow} or log P) >3 (Table 4). Thus, bioaccumulation is not a potential toxicological problem (see Section 5.2.1).

TABLE 4. LOG OCTANOL WATER PARTITION COEFFICIENTS OF THE ORGANIC CONTAMINANTS DETECTED IN WELL CC-33B

Contaminant	Log k_{ow}
Chlorobenzene	2.8[a]
Chloroform	1.9[b]
1,1,2-Trichloroethane	2.2[b]
1,1,1,2-Tetrachloroethane	3.0[c]
1,1,2,2-Tetrachloroethane	2.4[b]
1,1-Dichlorethene	2.1[a]
cis-1,2-Dichlorothene	1.8[d]
Trichloroethene	2.4[b]
Tetrachloroethene	2.9[b]
Vinyl chloride	1.4[c]

[a] Value taken from Howard (1989).
[b] Value taken from U.S. EPA (1991b).
[c] Value taken from Mackay et al. (1993).
[d] Value taken from Howard (1990).

5.1.6.2 Routine Water Quality Analyses

The raw data, mean, and standard deviation of the mean for each exposure tank in the chronic histopathology study are given in Burton et al. (1994). The methods for determining the routine water quality are also given in Burton et al. (1994). Some of the water quality parameters varied as a function of the three treatments; however, water quality within a given treatment was quite consistent over the 9-month study. The continuous temperature measurement recorded from Tank No. 3 during the 9-month study verified the constant temperature conditions which were measured daily in each exposure tank.

5.2 Hazard Assessment Evaluation

5.2.1 Toxicity

As discussed in Section 4.2, the ASTM guide recommends a phased hazard iteration process (ASTM, 1992b). Phase I (low-cost), Phase II (medium-cost), and Phase III (high-cost) toxicological data collection/iterations were all conducted simultaneously during the biomonitoring evaluation studies. Thus, the decision that normally would be made at the end of each phase concerning whether or not the potential hazard should be assessed as either 1) minimal, 2) potentially excessive, or 3) uncertain was not made until the end of the biomonitoring study when all of the toxicological studies were complete.

One important decision regarding bioaccumulation in aquatic organisms was made at the end of Phase I. The hazardous heavy metals present in the groundwater (Table 3) do not bioaccumulate to any degree in their metallic form in tissues (Williamson et al., 1993). Some heavy metals will bioaccumulate in the shells of benthic organisms; however, bioaccumulation in the shells is not believed to interfere with the normal physiological function of the organisms. With regard to the halogenated organics present in the groundwater, the log K_{ow}s (log P) of all the organics are ≤3 (Table 4). Bioaccumulation of a material up to 100-fold above background (bioconcentration factor or BCF = 100) can occur when the log K_{ow}s = 3 (U.S. EPA, 1991c). Thus, bioaccumulation was judged not to be an issue in the hazard assessment.

Toxicity was detected at various groundwater concentrations by all of the biomonitoring systems at pH 4 with the exception of two assays (Section 5.1). The Ames test for mutagenicity was negative in all cases including 100% groundwater as was the 9-month Japanese medaka chronic growth and histopathological test up to 10% groundwater by volume. A positive response was found in 100% groundwater in the SCE assay when the groundwater sample was concentrated ≈50,000X; the SCE assay was negative in 100% groundwater which was not concentrated. As discussed in Section 5.1.3.2, the SCE findings for concentrated 100% groundwater were not considered to be important to aquatic organisms in the near-field of the groundwater discharge.

The lowest concentration of groundwater that caused no observable adverse effect at pH 4, in the test systems in which the NOEC value could be determined, was 10% groundwater by volume. A NOEC of 10% groundwater by volume occurred in 3 out of 4 tests in both the cladoceran and FETAX assays; it occurred once in the fathead minnow test. The NOEC concentration was higher at pH 7 in both the fathead minnow and FETAX assays. The 10% groundwater by volume NOEC for cladoceran at pH 4, however, did not change when the organism was exposed to buffered groundwater at pH 7.

Maximum acceptable toxicant concentrations (MATC) are normally required for chronic toxicity estimates in a hazard assessment (ASTM, 1992b). The MATC is the highest concentration of a material that would have no statistically significant observed effect on the survival, growth, or reproduction of a test species during continuous exposure throughout a life-cycle or partial life cycle test. The short-term chronic tests with the green alga and cladoceran can be used to estimate the MATC for a plant and invertebrate (Weber et al., 1989). The green algal test is a multi-generation assay. The cladoceran test is a partial life cycle test. The MATC for the alga and cladoceran would be 18 and 10% groundwater by volume, respectively.

One can estimate chronic toxicity by calculating the acute-to-chronic ratio (ACR) which is simply the ratio of the LC50 to the NOEC (U.S. EPA, 1991a). The ACRs for the fathead minnow at pH 4 ranged from 1.4 to 2.0 in the four bimonthly tests. Thus, chronic toxicity of the groundwater to the fathead minnow would be estimated to be ≈2-fold greater than the acute toxicity. The estimated MATC of the groundwater to the fathead minnow would be ≈11% groundwater by volume. Although the 9-month Japanese medaka growth test is not a partial life cycle test by definition, the groundwater had no effect on Japanese medaka growth or chronic histopathology at 10% groundwater by volume after a 9-month exposure.

The estimated MATCs for the alga, invertebrate, and fish in the biomonitoring study suggest that the groundwater would not be harmful at a concentration of 10% groundwater by volume. Likewise, no genotoxicity, developmental toxicity, or chronic histopathology occurred at 10% groundwater by volume. Thus, the biomonitoring data suggest that chronic toxicity may not occur in the receiving stream at a dilution of 10:1. Chronic toxicity, however, would be predicted to occur in the Bush River if the groundwater entered the receiving stream at the concentrations which occur in well CC-33B.

In conclusion, if all of the conservative assumptions presented in Section 4.2 concerning the groundwater are assumed to occur (e.g., homogeneously mixed heavy metals and DNAPLs, no chemical or biotic transformations of the materials, large release volumes through the sediments to the Bush River, etc.), the groundwater would be considered a potentially excessive hazardous material to the benthic biota of the Bush River (ASTM, 1992b). The hazard to water column aquatic organisms would rapidly dissipate as the materials mixed in the receiving stream. We argue below that even if all of the conservative assumptions in Section 4.2 for the groundwater were used, that potential water quality impacts would be minimal if a mixing zone was granted by the State of Maryland which allows for local exceedences of water quality standards. The State of Maryland allows on a case-by-case basis a mixing zone where acute and chronic water quality criteria may be exceeded (see Section 5.2.4). Standards for a mixing zone for acute and chronic sediment quality do not currently exist.

5.2.2 Near-field Dilution in the Bush River

The results of the ULINE near-field model study are given in Najarian Associates, Inc. (1994). A copy of the Najarian report is included in this report as Appendix 2. The ULINE near-field model results indicate that a near-field dilution ratio of 159:1 is appropriate for application of the Maryland Department of the Environment's (MDE) acute aquatic life toxicity criteria. The model shows that a far-field dilution factor of 1,006:1 is

appropriate for application of chronic toxicity criteria. A dilution factor of 159:1 means that when 1 part of Beach Point groundwater is diluted in 159 parts of Bush River water, MDE's acute criteria are met at the edge of the acute mixing zone. MDE's acute and chronic aquatic life criteria are the same as those derived by EPA. Note that far-field flushing parameters are not included in the ULINE near-field model. When near-field dilution is corrected for the influence of far-field flushing, the resulting total near-field dilution ratios will decrease (see below).

A model sensitivity analysis was performed to evaluate the effects of various parameter variations on the near-field ULINE model output. The results of the model sensitivity analysis are given in Figures 2-6 of the Najarian report in Appendix 2. The sensitivity analysis showed that variations in groundwater discharge will change the dilution ratios. A groundwater discharge rate of 1,176 m^3/d (Section 4.2.3) gives a dilution ratio of 159. A higher discharge rate of 2,160 m^3/d, for example, will decrease the near-field ratio to 88. The length of the groundwater discharge in the Bush River affects the dilution ratio. The length of the discharge used in the model was assumed to be 448 m (entire length of Beach Point Peninsular). However, a more conservative estimate of approximately half the length of the peninsular (i.e., 224 m) gives a near-field dilution ratio of 81; a length of approximately one third gives a factor of 54. The sensitivity analysis also showed that dilution ratios are somewhat dependent upon the assumed discharge depth, are sensitive to ambient current speed, and relatively insensitive to salinity.

The total near-field dilutions (near-field dilution corrected for the influence of far-field dilution) for MDE's acute toxicity criteria under Spring and Fall conditions were 42 and 43, respectively. The total near-field dilutions for chronic toxicity criteria were 168 and 185, respectively, under Spring and Fall conditions (Appendix 2). (As discussed in Section 5.2.1 above, no acute or chronic toxicity occurred in any of the biomonitoring systems at a dilution of 10). Using the above total near-field dilution factors, near-field concentrations can be estimated for toxicants of interest. The estimate is obtained by dividing the groundwater contaminant concentration by the dilution factor. For example, using the Spring dilution factor of 42, a pollutant concentration of 42 µg/L would be diluted to 1 µg/L within the near-field mixing zone limit of the Beach Point discharge in the Bush River. Using this relationship, the near-field concentrations for all the groundwater contaminants can be calculated.

The computed acute and chronic dilutions for the highest concentration of each heavy metal and aliphatic organic measured in well CC-33B (Table 3) are given in Table 5. The computed

TABLE 5. COMPUTED DILUTIONS OF THE GROUNDWATER METALS AND ORGANICS IN THE BUSH RIVER[a]

Contaminant	Maximum Groundwater Concentration	Bush River Concentration	
		Acute[b]	Chronic[c]
Heavy Metals			
Aluminum	1,500	35.71	8.93
Arsenic	7	0.17	0.04
Boron	70	1.67	0.42
Cadmium	1.4	0.03	0.01
Cobalt	110	0.19	0.05
Copper	8	0.19	0.05
Nickel	180	4.29	1.07
Zinc	310	7.38	1.85
Volatile Organics			
Chlorobenzene	2	0.05	0.01
Chloroform	12	0.29	0.01
1,1,2-Trichloroethane	110	2.62	0.65
1,1,1,2-Tetrachloroethane	3	0.07	0.02
1,1,2,2-Tetrachloroethane	17,000	404.76	101.19
1,1-Dichlorethene	1	0.02	0.006
cis-1,2-Dichlorothene	640	15.24	3.81
Trichloroethene	1,600	38.10	9.52
Tetrachloroethene	90	2.14	0.54
Vinyl chloride	2	0.05	0.01

[a] All heavy metal and volatile organic concentrations are given as μg/L.
[b] Based on the maximum concentration measured during the four bimonthly chemical analyses and dilution ratio of 42:1 (see Section 5.2.2).
[c] Based on the maximum concentration measured during the four bimonthly chemical analyses and dilution ratio of 168:1 (see Section 5.2.2).

dilutions for both the acute and chronic values were determined using the Spring acute and chronic dilution factors (i.e., 42 and 168). An examination of the results in Table 5 show that the expected near-field impacts would be near or below the analytical method detection limit for most of the contaminants. For the heavy metals and organics that would be discharged in the Beach Point groundwater under the worst-case assumptions of this hazard assessment, the analysis suggests that on-site contamination will

not produce any discernable adverse effect on the water quality of the Bush River.

5.2.3 Mixing Zone Consideration

5.2.3.1 Mixing Zone Regulations

The State of Maryland allows for a mixing zone on a case-by-case basis. The State regulations regarding mixing zones are discussed in detail in the report by Najarian Associates, Inc. (1994). Briefly, as discussed in the Najarian report, a mixing zone is defined by the State of Maryland (COMAR 26.08.01.01B) as "an area contiguous to a discharge where surface water quality or groundwater quality does not have to met: (a) all water quality criteria, or (b) all requirements otherwise applicable to the natural water." That is, pollutant concentrations within the mixing zone may exceed the specified water quality standards within a localized vicinity of the outfall. The dimensions of the zone are normally comparable to those associated with the initial dilution process (e.g., 10-100 m). Thus, aquatic organisms would only be exposed to concentrations exceeding the specified criteria for a brief period during a transit of the mixing zone. COMAR 26.08.02.05A further states that mixing zones may be allowed only if they meet a number of requirements which ensure that no interference to aquatic ecosystem uses occurs outside the mixing zone (the list of requirements are given in Najarian Associates, Inc., 1994).

Four types of numerical toxic substance criteria are promulgated by the MDE: 1) human health criteria through ingestion of public water supplies; 2) the wholesomeness of fish for human consumption; 3) fresh, estuarine, and salt water aquatic life criteria from acute toxicity impacts; and 4) fresh, estuarine, and salt water aquatic life criteria from chronic toxicity impacts. For the purpose of applying numerical toxic substance criteria, the Bush River Area (Sub-Basin 02-13-07) is classified by MDE as "freshwater" (COMAR 26.08.02.03-1B) and as "Use II" (shellfish harvesting waters). The toxic substances criteria for both ambient surface waters ("freshwater") and human health ("fish consumption") for the pollutants in well CC-33B at Beach Point are given in Table 6. Toxic criteria related to drinking water do not apply to shellfish harvesting waters.

The following regulations (COMAR 26.08.02.05D) pertain to the application of toxic substance chronic criteria for the protection of aquatic life. First, in terms of size, the Regulatory Mixing Zone (i.e., the mixing zone in which the chronic criteria are applied) may not exceed 10 percent of the cross-sectional area of the receiving waters. Also, the chronic criteria are applied under flow conditions determined from site-specific data for the mean tide level, average tidal velocity,

TABLE 6. APPLICABLE TOXIC SUBSTANCES CRITERIA FOR THE BUSH RIVER AND A COMPARISON OF PROJECTED NEAR-FIELD GROUNDWATER CONCENTRATIONS[a]

Constituent	Acute Conditions		Chronic Conditions		
	Aquatic Life Criteria	Near-Field Conc.	Aquatic Life Criteria	Near-Field Conc.	Human Health Fish Consumption
Heavy Metals					
Aluminum	[b]	35.71	[b]	8.93	[c]
Arsenic (III)	360[d]	0.17(tot)	190[d]	0.04 (tot)	[c]
Cadmium	3.9[d]	0.03	1.1[d]	0.01	[c]
Copper	18[d]	0.19	12[d]	0.05	[c]
Nickel	1,400[d]	4.29	160[d]	1.07	100
Zinc	120[d]	7.38	110[d]	1.85	[c]
Volatile Organics					
Chlorobenzene	[e]	0.05	[e]	0.01	[c]
Chloroform	28,900[f]	0.29	1,240[f]	0.01	[c]
1,1,2-Trichloroethane	[e]	2.62	9,400[f]	0.65	[c]
1,1,1,2-Tetrachloroethane	[e]	0.07	[e]	0.02	[c]
1,1,2,2-Tetrachloroethane	[e]	404.76	2,400[f]	101.19	[c]
1,1-Dichlorethene	[e]	0.02	[e]	0.006	[c]
cis-1,2-Dichlorothene	[e]	15.24	[e]	3.81	[c]
Trichloroethene	[e]	38.10	[e]	9.52	807
Tetrachloroethene	[e]	2.14	[e]	0.54	[c]
Vinyl chloride	[e]	0.05	[e]	0.01	[c]

[a] All heavy metal and volatile organic concentrations are given as μg/L.
[a] EPA acute and chronic aquatic life criteria are pH dependent; State of Maryland has no standard for aluminum.
[b] No State of Maryland criteria.
[c] Acute and chronic aquatic life criteria are hardness dependent (100 mg/L $CaCO_3$ used).
[d] No State of Maryland or EPA criteria.
[e] No State of Maryland or EPA criteria. Value presented is EPA's LOEC.

and, when appropriate, the design stream flow. Based on MDE's discretion, a plume dilution study may also be required for the application of these criteria. In accordance with COMAR 26.08.02.06(a), conditions pertinent to the application of toxic substance chronic criteria are also relevant to the application of toxic substance human health criteria.

The following regulations (COMAR 26.08.02.05C) pertain to the application of toxic substance acute criteria for the protection of aquatic life. Such acute criteria (for low velocity dischargers) must be meet within a short distance from the outfall using the most restrictive of the following:

1) Within 10 percent of the distance (in any spatial direction) from the discharge to the edge of the Regulatory Mixing Zone used for the application of the chronic criteria.

2) Within a distance of 50 times at the "discharge length scale" (i.e., 50 times the square root of the cross-sectional area of the discharge outlet).

3) Within a distance of 5 times the local water depth in any horizontal direction from the discharge outlet.

Thus, the region where the acute criteria are applied, sometimes called the "Toxic Dilution Zone", extends outward from the discharge to a distance no larger than any of the three lengths cited above. Moreover, the Toxic Dilution Zone must not occupy more than 5 percent of the cross-sectional area of the receiving waters. Finally, the acute criteria are applied under flow conditions determined from site-specific data for the mean low water elevation, the minimum daily averaged 1-h tidal velocity and, when appropriate, design stream flow conditions. Such tidal stages are deemed "critical" (minimum dilution) conditions.

5.2.3.2 Site Applicability

The regulatory conditions relevant to mixing zones were developed for outfall diffusers. Their application to an area-wide groundwater discharge is somewhat uncertain due to the unknown extent of the area. It seems likely that the discharge area is confined within the Regulatory Mixing Zone, which for Beach Point, extends approximately 150 m (≈500 ft) from the river bank. However, the extent of the Toxic Dilution Zone is more poorly defined because it partially depends on the configuration of the groundwater discharge. Thus, the applicability of the mixing zone concept must be further examined.

There are three approaches used for the assessment of toxic substance discharges in the United States (U.S. EPA, 1993b). The concept of a mixing zone provides an intermediate approach to the assessment of toxic pollutant impacts. The two other approaches

generally in use are "end of pipe" and "fully mixed". The "end of pipe" approach allows for zero dilution; aquatic life criteria are applied directly to the discharge water quality. This simplified approach yields the most stringent results which, in turn, most ensures the protection of aquatic biota. However, as Najarian Associates, Inc. (1994) points out, it may lack a physical or biological basis in terms of receiving water impacts. Thus, its results are not considered to be defensible. It is effectively a non-technically based policy decision.

Conversely, the "fully mixed" (or far-field) approach considers the total volume of a receiving water segment to be instantly available for the dilution of a discharge. This approach is generally far less stringent than the "end of pipe" approach. However, it may not adequately address biological impacts in the receiving water due to near-field conditions. The concept of mixing zones evolved to address these limitations. Mixing zones allow consideration of the initial dilution process. This approach also protects aquatic life by limiting the potential for exposure to elevated concentrations to minor portions of the waterway. Since only a portion of the available dilution is considered in this approach, mixing zones provide a much more stringent basis for regulation than the "fully mixed" approach.

In the application of the mixing zone concept to the Bush River, it was necessary to conceptualize the area-wide groundwater discharge from Beach Point as a line source (diffuser). Because of this approach, potential impacts could be assessed in a technically defensible manner. However, the approach had a side effect of concentrating the groundwater impact into a smaller area. That is, the approach minimized the available near-field dilution and projected higher receiving water concentrations than would actually occur. Thus, the mixing zone analysis produced an overly conservative result. In the near-field, actual receiving water concentrations would be less than those projected due to the more dispersed nature of the groundwater plume. In the near-field, projected and actual receiving water concentrations should be comparable. However, far-field concentrations would always be less than the near-field projections.

5.2.3.3 Conformance with Aquatic Life Criteria

The anticipated near-field receiving water concentrations are compared in Table 6 with the water quality criteria for the contaminants in well CC-33B. As can be seen in Table 6, no exceedences of the criteria are projected. It is important to point out that no exceedences of the criteria are projected for groundwater discharge pollutants which were assumed to have the highest concentrations measured in well CC-33B. The assumption that the groundwater discharge would have concentrations as high as those in well CC-33B is very conservative.

There would be detectable concentrations of some of the chlorinated organics in the receiving stream if the contaminant distribution assumptions in Section 4.2.4 are used in the near-field projection. With the exception of trichloroethene for human health fish consumption, the State of Maryland has not implemented criteria for the remaining organics present in the groundwater.

5.2.3.4 Re-evaluation of Hazard Assessment Assumptions as an Alternative to a Mixing Zone

It is argued above that the State of Maryland could consider a mixing zone for the discharge of Beach Point groundwater to the Bush River. The argument for a mixing zone, however, is based on the assumption that rather large volumes of groundwater with high concentrations of contaminants may enter the Bush River. As discussed in Sections 4.2.3 and 4.2.4, the volume of groundwater and the concentrations of contaminants in the groundwater that were assumed to occur in the near-field mixing zone were based on a number of very conservative assumptions. It is highly unlikely that the contaminant loads assumed in the near-field dilution model will ever occur in the Bush River. Thus, the assumptions that were used in the dilution model should be evaluated further before a decision is made to request a mixing zone. A refinement of the assumptions would reduce the volume of the discharge as well as the concentration of contaminants that may enter the receiving stream. Finally, the State may not consider a mixing zone for a groundwater discharge as a policy to reduce contaminant loading in the Bush River.

A better estimate of the discharge rate of the surficial aquifer is needed. Several of the conservative assumptions discussed in Section 4.2.3 concerning the discharge rate could be re-evaluated to obtain better estimates that could be used in the discharge rate calculation. The following could be examined: 1) Is the horizontal transport value V_{hor} of 0.27 m/d (0.89 ft/d), which was taken from one 25-h study in an intermediate depth well (CC-33B.1), representative of the entire aquifer? 2) Is the aquifer homogeneous? 3) Does isotropic flow occur throughout the aquifer? 4) Is the average saturated thickness of the aquifer 18.8 m (61.8 ft)? 5) Does the aquifer discharge over the entire length of Beach Point Peninsula fronting the Bush River?

The following assumptions concerning the concentrations of contaminants in the groundwater (Section 4.2.4) may be considered: 1) What is the extent of heavy metal retardation via adsorption onto solid surfaces or trapping by clays through ion exchange? 2) How important is the precipitation of metals when the pH shifts from 4 to neutrality as the groundwater moves through the sediments into the receiving stream? 3) How are the DNAPL concentrations changed as abiotic (chemical) and biotic (microbial) transformations occur? 4) The assumption was made that the heavy metals and residual DNAPLs were homogeneously

mixed throughout the complete aquifer. The concentrations of the materials in the shallow and intermediate wells clearly show that this is not the case. Thus, better estimates of the distribution (concentration) of the materials in the aquifer are needed. This is particularly important for vertical flux estimates of the DNAPLs.

SECTION 6

CONCLUSIONS

6.1 Biomonitoring Evaluation

Several EPA priority pollutant heavy metals were found in the groundwater. The concentrations of cadmium, copper, nickel, and zinc exceeded one or more of EPA's numerical water quality criteria for the specific metal. A number of chlorinated aliphatic compounds were also found in the groundwater, several of which are EPA priority pollutants. None of the priority pollutant organics found in the groundwater have EPA numerical water quality criteria values because insufficient data exist to develop criteria. 1,1,2,2-Tetrachlorethane concentrations found in the groundwater exceeded EPA's freshwater chronic LOEC value. None of the organics found in the groundwater have octanol water partition coefficients (log k_{ow} or log P) >3; thus, bioaccumulation is not a potential toxicological problem.

Toxicity was detected at various groundwater concentrations by all biomonitoring systems at pH 4 with the exception of two assays. The Ames assay for mutagenicity was negative in all cases (1, 10, and 100% groundwater by volume). Japanese medaka growth was not affected by 9 months of exposure to 1 and 10% groundwater by volume in the chronic histopathology assay. No significant lesions were found in the Japanese medaka exposed to groundwater concentrations up to 10% groundwater by volume. A positive response was found for 100% groundwater in the SCE assay when the groundwater sample was concentrated ≈50,000X. The SCE assay was negative for unconcentrated 100% groundwater. The positive SCE response in the 50,000X concentrated sample was judged not to be important to aquatic organisms in the vicinity of the groundwater discharge.

The lowest concentration of groundwater that caused no observable adverse effect at pH 4, in the test systems in which the NOEC value could be determined, was 10% groundwater by volume. A NOEC of 10% groundwater by volume occurred in 3 out of 4 tests in both the cladoceran and FETAX assays; it occurred once in the fathead minnow test. The NOEC concentration was higher at pH 7 in both the fathead minnow and FETAX assays. The 10% groundwater by volume NOEC for cladoceran at pH 4, however, did not change when the organism was exposed to buffered groundwater at pH 7.

The data for the alga, invertebrate, and fish in the biomonitoring study suggest that the groundwater would not be harmful at a concentration of 10% groundwater by volume. Likewise, no genotoxicity, developmental toxicity, or chronic histopathology occurred at 10% groundwater by volume. Thus, the biomonitoring results suggest that chronic toxicity may not occur

in the receiving stream at a dilution of 10:1. Chronic toxicity would be predicted to occur in the Bush River if the groundwater entered the receiving stream at the concentrations which occur in well CC-33B.

6.2 Hazard Assessment Evaluation

The groundwater may be considered a potentially hazardous material to the benthic biota of the Bush River when conservative groundwater assumptions are used (e.g., homogeneously mixed heavy metals and chlorinated organics, no chemical or biotic transformations of the DNAPLs, large release volumes through the sediments to the Bush River, etc.). The hazard to water column aquatic biota would rapidly dissipate as the groundwater materials are mixed in the receiving stream. Because the potential water quality impacts were judged to be minimal, a mixing zone approach by the State of Maryland which allows for local exceedences of water quality standards may be pursued.

The near-field (ULINE model) and far-field (dye-tracer model) screening level dilution models suggested that a total near-field dilution of approximately 42:1 for the application of Maryland's acute aquatic life criteria and a near-field dilution level of 168:1 for the application of chronic criteria would occur in the Bush River. Thus, contaminants introduced via Beach Point groundwater into Bush River receiving waters at a concentration of 42 µg/L would be diluted locally to a concentration of approximately 1 µg/L or less. When the dilution factors were applied to groundwater quality at Beach Point, none of the heavy metals or chlorinated aliphatic compounds exceeded Maryland's current acute or chronic aquatic life criteria. The dilution study showed that detectable concentrations of some of the chlorinated organics would occur in the receiving stream when the conservative assumptions concerning the groundwater contaminants were used in the model.

Although an argument can be made for a mixing zone, it is highly unlikely that the contaminant loads assumed in the near-field dilution model will ever occur in the Bush River. The conservative assumptions that were used in the hazard assessment should be evaluated further before a decision is made that the groundwater entering the Bush River is an environmental hazard and, thus, a mixing zone should be considered. A refinement of the assumptions would reduce the uncertainty regarding the volume of the discharge as well as the concentration of contaminants that may enter the receiving stream.

SECTION 7

REFERENCES

Ames, B.N., W.E. Durston, E. Yamasaki, and F.D. Lee. 1973. Carcinogens are mutagens: A single test system combining liver homogenate for activation and bacteria for detection. Proc. Natl. Acad. Sci. (USA) 70:2281-2286.

APHA et al. (American Public Heath Association, American Water Works Association, and Water Pollution Control Federation). 1989. Standard methods for the examination of water and wastewater, 17th ed. Amer. Public Hlth. Assoc., Washington, DC.

ASTM (American Society for Testing and Materials). 1992a. Standard guide for conducting the frog embryo teratogenesis assay-Xenopus (FETAX). ASTM Designation E 1439-91. Pages 1199-1209 in: 1992 Annual book of ASTM standards, Vol. 11.04. Amer. Soc. Testing Materials, Philadelphia.

ASTM (American Society for Testing and Materials). 1992b. Standard guide for assessing the hazard of a material to aquatic organisms and their uses. ASTM Designation E 1023-84 (Reapproved 1988). Pages 669-684 in: 1992 Annual book of ASTM standards, Vol. 11.04. Amer. Soc. Testing Materials, Philadelphia.

Baker, J.P., D.P. Bernard, S.W. Christensen, M.J. Safe, J. Freda, K.J. Heltcher, D.R. Marmorek, L. Rowe, P.F. Scanlon, G.W. Suter II, W.J. Warren-Hicks, and P.M. Welbourn. 1990. Biological effects of changes in surface water acid-base chemistry. NAPAP Report 13. in: Acidic deposition: State of science and technology, Vol. II Aquatic processes and effects. National Acid Precipitation Assessment Program, U.S. Environmental Protection Agency, Washington, DC.

Bantle, J.A. 1994. Personal communication. Oklahoma State Univ., Stillwater, OK.

Bantle, J.A., J.N. Dumont, R.A. Finch, and G. Linder. 1991. Atlas of abnormalities a guide for the performance of FETAX. Oklahoma State Univ., Stillwater, OK.

Barnhouse, L.W., G.W. Suter, S.M. Bartell, J.J. Beauchamp, R.H.Gardner, R.V. O'Neill, and A.E. Rosen. 1986. Users manual for ecological risk assessment. ORNL-6251. Oak Ridge National Lab, Oak Ridge, TN.

Bunton, T.E. 1990. Hepatopathology of diethylnitrosamine in the medaka (Oryzias latipes) following short-term exposure. Toxicol. Pathol. 18:313-323.

Burton, D.T., R.S. Heriott, and S.D. Turley. 1994. Toxicity and hazard assessment evaluation of contaminated groundwater at Aberdeen Proving Ground-Edgewood Area Beach Point Peninsula. WREC-94-B3. Univ. Maryland, Wye Research and Education Center, Queenstown, MD.

Cairns, J., Jr. (ed.). 1986. Community toxicity testing. ASTM STP 920. Amer. Soc. Testing Materials, Philadelphia, PA.

Cairns, J., Jr. 1990. The genesis of biomonitoring in aquatic ecosystems. Environ. Professional 12:169-176.

Carter, H.H. 1976. Simple one dimensional kinematic model results for the Bush River and Romney Creek. Spec. Rep. 49. Johns Hopkins Univ. Chesapeake Bay Inst., Baltimore.

Cohen, R.M. and J.W. Mercer. 1993. DNAPL site evaluation. CRC Press, Boca Raton, FL.

Dawson, D.A., C.A. McCormick, and J.A. Bantle. 1985. Detection of teratogenic substances in acidic mine water samples using the frog embryo teratogenesis assay-Xenopus (FETAX). J. Appl. Toxicol. 5:234-244.

de March, B.G.E. 1988. Acute toxicity of binary mixtures of five cations (Cu^{2+}, Cd^{2+}, Zn^{2+}, Mg^{2+}, and K^{+}) to the freshwater amphipod Gammarus lacustris (Sars): Alternative description models. Can. J. Fish. Aquat. Sci. 45:625-633.

Dickson, K.L., A.W. Maki, and J.Cairns, Jr. (eds.). 1979. Analyzing the hazard evaluation process. Amer. Fish. Soc., Bethesda, MD.

Dutka, B.J. and K.K. Kwan. 1988. Battery of screening tests approach applied to sediment extracts. Toxicol. Assessment 3:303-314.

Experimental Pathology Laboratories, Inc. 1994. U.S. Army Biomedical Research and Development Laboratory test 410-001R Beach Point groundwater study with medaka, Vol. 1-3. EPL Project No. 406-027. Exp. Pathology Lab., Inc., Herndon, VA.

Faust, M., R. Altenburger, W. Boedeker, and L.H. Grimme. 1994. Algal toxicity of binary combinations of pesticides. Bull. Environ. Contam. Toxicol. 53:134-141.

Fisher, H.B., E.J. List, R.C.Y. Koh, J. Imberger, and N.H. Brooks. 1979. Mixing in inland and coastal waters. Academic Press, New York.

Goodman, L.R. 1986. Proposed new standard guide for conducting early life-stage toxicity tests with fishes. Draft No. 10. Amer. Soc. Testing Materials, Philadelphia.

Gulley, D.H., A.M. Boelter, and H.L. Bergman. 1989. Toxstat release 3.0. Univ. Wyoming, Laramie, WY.

Hawkins, W.E. 1991. Development of carcinogenesis bioassay models: Response of small fish species to various classes of carcinogens. Contract No. DAMD17-88-C-8050. Gulf Coast Res. Lab., Ocean Springs, MS.

Hawkins, W.E. 1994. Personal communication. Gulf Coast Res. Lab., Ocean Springs, MS.

Hawkins, W.E., R.M. Overstreet, and W.W. Walker. 1988. Small fish models for identifying carcinogens in the aqueous environment. Water Resour. Bull. 24:941-949.

Herricks, E.E. and D.J. Schaffer. 1985. Can we optimize biomonitoring? Environ. Management 9:487-492.

Herriott, R.S. and D.T. Burton. 1992. U.S. Army Biomedical Research and Development Laboratory aquatic biomonitoring trailer version 1.0: Operations manual. AD A260733. Defense Tech. Information Center, Alexander, VA.

Hinton, D.E., J.A. Hampton, and P.A. McCuskey. 1985. Japanese medaka liver tumor model: Review of literature and new findings. Pages 439-450 in: Jolley, R.L, R.J. Bull, W.P. Davis, S. Kata, M.H. Roberts, Jr., and V.A. Jacobs, eds. Water chlorination chemistry, environmental impact and health effects, Vol. 5. Lewis Publ., Inc., Ann Arbor, MI.

Hinton, D.E., J.A. Couch, S.J. Teh, and L.A. Courtney. 1988. Cytological changes during progression of neoplasia in selected fish species. Aquatic Toxicol. 11:77-112.

Hoffman, G.R. 1991. Genetic toxicology. Pages 201-225 in: Amdur, M.O., J. Doull, and C.D. Klaassen, eds. Casarett and Doull's toxicology the basic science of poisons, 4th ed. Pergamon Press, New York.

Horning, W.B. and C.I Weber (eds.). 1985. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms. EPA/600/4-85/014. U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Cincinnati, OH.

Howard, P.H. (ed.). 1989. Handbook of environmental fate and exposure data for organic chemicals, Vol. 1 Large production and priority pollutants. Lewis Publ., Inc., Chelsea, MI.

Howard, P.H. (ed.). 1990. Handbook of environmental fate and exposure data for organic chemicals, Vol. 2 Solvents. Lewis Publ., Inc., Chelsea, MI.

Hughes, J.S., M.M. Alexander, and K. Balu. 1988. An evaluation of appropriate expressions of toxicity in aquatic plant bioassays as demonstrated by the effects of atrazine on algae and duckweed. Pages 531-547 in: Adams, W.J., G.A. Chapman, and W.G. Landis, eds. Aquatic toxicology and environmental fate, Vol. 10. ASTM STP 971. Amer. Soc. Testing Materials, Philadelphia.

Ishikawa, T. and S. Takayama. 1979. Importance of hepatic neoplasms in lower vertebrate animals as a tool in cancer research. J. Toxicol. Environ. Hlth. 5:537-550.

JEG (Jacobs Engineering Group, Inc.). 1993. Beach Point test site Aberdeen Proving Ground - Edgewood Area, Maryland focused feasibility study final project work plan. Master Agreement No. 071914-A-D7, Task Order No. 142133, Supplement No. 8. Jacobs Engineering Group, Inc., Washington, DC.

JEG (Jacobs Engineering Group, Inc.). 1994. Estimated discharge rates in the surficial aquifer Beach Point (Peninsula) study area Aberdeen Proving Ground-Edgewood Area, Maryland. Memorandum by Scott Treherne, May 23, 1994. No document reference number. Jacobs Engineering Group, Inc., Washington, DC.

Kaiser, K.L.E. and J.M. Ribo. 1988. Photobacterium phosphoreum toxicity bioassay. II. Toxicity data compilation. Toxicity Assessment: An Internat. J. 3:195-237.

Klaunig, J.E., B.A. Barut, and P.J. Goldblatt. 1984. Preliminary studies on the usefulness of medaka, Oryzias latipes, embryos in carcinogenicity testing. Natl. Cancer Inst. Monogr. 65:155-161.

Knox, R.C., D.A. Sabatini, and L.W. Canter. 1993. Subsurface transport and fate processes. Lewis Publ., Ann Arbor, MI.

K-V Associates, Inc. 1994. Hydrogeologic study of Beach Point, Edgewood Area Aberdeen Proving Grounds, Maryland. Project No. 1016. K-V Associates, Inc., Falmouth, MA.

Lauren, D.J., S.J. Teh, and D.E. Hinton. 1990. Cytotoxicity phase of diethylnitrosamine-induced hepatic neoplasia in medaka. Cancer Res. 50:5504-5514.

Lee, G.F. 1973. Chemical aspects of bioassay techniques for establishing water quality criteria. Water Res. 7:1525-1546.

Lorah, M.M. 1994. Personal communication. U.S. Geological Survey, Towson, MD.

Lorah, M.M. and J.S. Clark. 1992. Contamination of ground water, surface water, and soil and evaluation of selected pumpage scenarios in the Canal Creek Area of Aberdeen Proving Ground, Maryland. Draft Open File Report 92-. U.S. Geological Survey, Towson, MD.

Lorah, M.M. and D.A. Vroblesky. 1989. Inorganic and organic ground-water chemistry in the Canal Creek Area of Aberdeen Proving Ground, Maryland. USGS Water-Resources Investigations Report 89-4002. U.S. Geological Survey, Towson, MD.

Mackay, D.W., P.J. Holmes, and C.J. Redshaw. 1989. The application of bioassay techniques to water pollution problems - The United Kingdom experience. Hydrobiologia 188/189:77-86.

Mackay, D., W.Y. Shiu, and K.C. Ma. 1993. Illustrated handbook of physical-chemical properties and environmental fate for organic chemicals, Vol. III Volatile organic chemicals. Lewis Publ., Inc., Ann Arbor, MI.

Marking, L.L. 1985. Toxicity of chemical mixtures. Pages 164-176 in: Rand, G.M. and S.R. Petrocelli, eds. Fundamentals of aquatic toxicology methods and applications. Hemisphere Publ. Corp., New York.

McGinnis, L.D., C.R. Daudt, M.D. Thompson, S.F. Miller, W.A. Mandell, and J. Wrobel. 1994. Environmental geophysics at Beach Point Aberdeen Proving Ground, Maryland. No document reference number. Argonne National Laboratory, Argonne, IL.

Metcalfe, C.D. 1989. Tests for predicting carcinogenicity in fish. Rev. Aquatic Sci. 1:111-129.

Microtox®. 1988. Microtox® Model 500 Toxicity Test System Manual. Microbics Corp., Carlsbad, CA.

Miller, W.E., J.C. Greene, and T. Shiroyama. 1978. The Selenastrum capricornutum Printz algal assay bottle test. Experimental design, application and data interpretation protocol. EPA-600/9-78-018. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Muellenhoff, W.P., A.M. Soldate, D.J. Baumgartner, M.D. Schuldt, L.R. Davis, and W.E. Frick. 1985. Initial mixing characteristics of municipal ocean discharges: Vol. 2 Procedures and applications. EPA 600/3-85-073a. U.S. Environmental Protection Agency, Environmental Research Laboratory, Narragansett, RI.

Najarian Associates, Inc. 1994. Screening-level assessment of the near-field dilution of groundwater discharges from the Beach Point Site. Job 1437. Najarian Assoc., Inc., Eatontown, NJ.

National Research Council. 1981. Testing for effects of chemicals on ecosystems. National Academy Press, Washington, DC.

Nemath, G. 1989. RCRA facility assessment report Edgewood Area Aberdeen Proving Ground, Maryland. Rep. No. 39-26-0490-90. U.S. Army Environ. Hygiene Agency, Aberdeen Proving Ground, MD.

Neuhold, J.M. 1986. Toward a meaningful interaction between ecology and aquatic toxicology. Pages 11-21 in: Poston, T.M. and R. Purdy, eds. Aquatic toxicology and environmental fate, Vol. 10. ASTM STP 921. Amer. Soc. Testing Materials, Philadelphia.

Oliveros, J.P. and D.A. Vroblesky. 1989. Hydrogeology of the Canal Creek Area, Aberdeen Proving Ground, Maryland. USGS Water-Resources Investigations Report 89-4021. U.S. Geological Survey, Towson, MD.

Peltier, W.H. and C.I. Weber. 1985. Methods for measuring the acute toxicity of effluents to freshwater and marine organisms, 3rd ed. EPA/600/4-85/013. U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Cincinnati, OH.

Perry, P.E. and H.J. Evans. 1975. Cytological detection of mutagens/carcinogens exposure by sister chromatid exchange. Nature 258:121-125.

Pritchard, D.W. 1976. Distribution of contaminants and excess heat in the Bush River and Romney Creek from the proposed Perryman Electric Power Plant. Spec. Rep. 58. Johns Hopkins Univ. Chesapeake Bay Inst., Baltimore.

Potts, K.W. 1994. Personal communication. U.S. Environmental Protection Agency, Office of Water, Heath and Ecological Criteria, Water Quality Criteria, Washington, DC.

Qureshi, A.A., K.W. Flood, S.R. Thompson, S.M. Janhurst, C.S. Inniss, and D.A. Rokosh. 1982. Comparison of a luminescent bacterial test with other bioassays for determining toxicity of pure compounds and complex effluents. Pages 179-195 in: Pearson, J.G., R.B. Foster, and W.E. Bishop, eds. Aquatic toxicology and hazard assessment, Vol. 5. ASTM STP 776. Amer. Soc. Testing Materials, Philadelphia.

Rand, G.M. and S.R. Petrocelli (eds.). 1985. Fundamentals of aquatic toxicology methods and applications. Hemisphere Publ. Corp., New York.

SAS. 1987. SAS/STAT™ guide for personal computer, 6th ed. SAS Inst., Inc., Cary, NC.

Schaeffer, D.J. and K.G. Janardan. 1987. Designing batteries of short-term tests with largest inter-tier correlation. Ecotoxicol. Environ. Safety 13:316-323.

Sigma Stat. 1992. Sigma Stat statistical software for working scientists, version 1.01. Jandell Scientific, Inc., San Rafael, CA.

Sprague, J.B. 1985. Factors that modify toxicity. Pages 123-163 in: Rand, G.M. and S.R. Petrocelli, eds. Fundamentals of aquatic toxicology methods and applications. Hemisphere Publ. Corp., New York.

Stephen, C.E. 1978. EPA statistics program, version 1.4. U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, MN.

Suter, G.W., II. 1993a. Organism-level effects. Pages 175-246 in: Suter, G.W., II, ed. Ecological risk assessment. Lewis Publ., Ann Arbor, MI.

Thurston, R.V., R.C. Russo, and K. Emerson. 1979. Aqueous ammonia equilibrium - Tabulation of percent ionized ammonia. EPA-600/3-79-091. U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, MN.

U.S. Biomedical Research and Development Laboratory. 1993. SOP for the determination of HMX, RDX, TNB, Tetryl, TNT, 2,6-DNT, and 2,4-DNT in water by high performance liquid chromatography (HPLC). No SOP Number. U.S. Biomedical Research and Development Laboratory, Ft. Detrick, Frederick, MD.

U.S. EPA. 1980a. Ambient water quality criteria for nickel. EPA 440/5-80-060. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1980b. Ambient water quality criteria for chlorinated ethanes. EPA 440/5-80-029. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1984a. Ambient water quality criteria for cadmium - 1984. EPA 440/5-84-032. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1984b. Ambient water quality criteria for copper - 1984. EPA 440/5-84-031. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1985a. Toxic substances control act test guidelines; final rules. Federal Register 50:39252-39516.

U.S. EPA. 1985b. Water quality assessment: A screening procedure for toxic and conventional pollutants. EPA/600/6-85/002b. U.S. Environmental Protection Agency, Athens, GA.

U.S. EPA. 1986a. Revision of toxic substances control act test guidelines. Federal Register 51:1522-1543.

U.S. EPA. 1986b. Quality criteria for water 1986. EPA/5-86-001. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1987. Ambient water quality criteria for zinc - 1987. EPA 440/5-87-003. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1991a. Technical support document for water quality-based toxics control. EPA/505/2-90-001. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1991b. Evaluation of dredged material proposed for ocean disposal. EPA-503/8-91/001. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1991c. Assessment and control of bioconcentratable contaminants in surface waters. 1991 Draft Report. U.S Environmental Protection Agency, Washington, DC.

U.S. EPA. 1993a. Technical basis for deriving sediment quality criteria for nonionic organic contaminants for the protection of benthic organisms by using equilibrium partitioning. EPA-822-R-93-011. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1993b. Water quality standards handbook, 2nd ed. EPA-823-B-93-002. U.S. Environmental Protection Agency, Washington, DC.

Walker, W.W., C.S. Manning, R.R. Overstreet, and W.E. Hawkins. 1985. Development of aquarium fish models for environmental carcinogenesis: an intermittent-flow exposure system for volatile hydrophobic chemicals. J. Appl. Toxicol. 5:255-260.

Weber, C.I. (ed.). 1991. Methods for measuring the acute toxicity of effluents and receiving waters to freshwater and marine organisms, 4th ed. EPA/600/4-90/027. U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Cincinnati, OH.

Weber, C.I., W.H. Peltier, T.J. Norberg-King, W.B. Horning, II, F.A. Kessler, J.R. Menkedick, T.W. Neiheisel, P.A. Lewis, D.J. Klemm, Q.H. Pickering, E.L. Robinson, J.M. Lazorchak, L.J. Wymer, and R.W. Freyberg. 1989. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms, 2nd ed. EPA/600/4-89/001. U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Cincinnati, OH.

Weis, J.S. and P. Weis. 1989. Effects of environmental pollutants on early fish development. Rev. Aquat. Sci. 1:45-73.

Williamson, R.L., D.T. Burton, J.H. Clarke, and L.E. Fleming. 1993. Gathering danger: The urgent need to regulate substances that can bioaccumulate. Ecol. Law Quart. 20:605-720.

Winner, R.W. 1989. Multigeneration life-span tests of the nutritional adequacy of several diets and culture waters for Ceriodaphnia dubia. Environ. Toxicol. Chem. 8:513-520.

APPENDIX 1

COMPREHENSIVE CHEMICAL ANALYSES CONDUCTED ON RAW (pH ≈4) BEACH POINT GROUNDWATER (WELL CC-33B) AND CHRONIC HISTOPATHOLOGY EXPOSURE TANKS

TABLE A1-1. RESULTS OF THE APG-EA BEACH POINT GROUNDWATER AND DILUENT WATER COMPREHENSIVE CHEMICAL ANALYSES (TEST NO. 1) - GENERAL WATER QUALITY[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Total Alkalinity ($CaCO_3$)	EPA 310.1	5	[b]	21	28	26
Ammonia (N)	EPA 350.1	0.02	0.029	0.025	0.029	0.027
Chloride (Cl)	EPA 300.0	0.5	490	86.7	31.2	21.9
Total Cyanide (CN)	EPA 335.2	10	ND	ND	ND	ND
Fluoride (F)	EPA 300.0	0.2	1.2	0.66	0.55	0.84
Hardness, ($CaCO_3$)	SM2540B		170	73	62	59
Nitrate (N)	EPA 300.0	0.1	0.14	1.5	1.7	4.3
Nitrite (N)	[b]	[b]	[b]	[b]	[b]	[b]
Total Phosphate (P)	EPA 365.4	0.01	2.64	3.13	0.26	0.54
Total Suspended Solids	EPA 160.2	5	ND	ND	ND	ND
Specific Conductance	EPA 120.1		1600	265	175	169
Sulfate (SO_4)	EPA 300.0	0.5	91.2	34.2	29.3	28.1
Sulfide (S)	EPA 376.1	5	180	350	360	330
TOC	EPA 415.2	1	4	2	ND	ND
Bromide (Br)	EPA 300.0	0.1	1.5	0.16	ND	ND

[a] All results expressed as mg/L except for specific conductance which is expressed as μmohs/cm.

[b] Analysis not conducted by Vendor.

TABLE A1-1. (CONTINUED) - METALS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Aluminum (Al)	EPA 200.7	40	810	64	ND	ND
Antimony (Sb)	EPA 200.7	20	ND	ND	ND	ND
Arsenic (As)	EPA 206.2	5	ND	ND	ND	ND
Beryllium (Be)	EPA 200.7	5	ND	ND	ND	ND
Boron (B)	EPA 200.7	50	54	51	ND	ND
Cadmium (Cd)	EPA 213.2	0.5	0.76	ND	ND	ND
Calcium (Ca)	EPA 200.7	500	16500	17300	17200	16700
Chromium (Cr)	EPA 200.7	5	ND	ND	ND	ND
Cobalt (Co)	EPA 200.7	40	110	ND	ND	ND
Copper (Cu)	EPA 200.7	5	5.4	ND	ND	ND
Iron (Fe)	EPA 200.7	100	4600	340	ND	ND
Lead (Pb)	EPA 239.2	5	ND	ND	ND	ND
Magnesium (Mg)	EPA 200.7	500	31300	7290	4590	4230
Manganese (Mn)	EPA 200.7	15	1860	200	22	ND
Mercury (Hg)	EPA 245.1	0.5	ND	ND	ND	ND
Molybdenum (Mo)	EPA 200.7	40	ND	ND	ND	ND
Nickel (Ni)	EPA 200.7	8	120	13	ND	ND
Potassium (K)	EPA 258.1	100	4130	2050	1760	1710
Selenium (Se)	EPA 270.2	5	ND	ND	ND	ND
Silver (Ag)	EPA 272.2	0.5	ND	ND	ND	ND
Sodium (Na)	EPA 200.7	500	257000	39200	12400	9260
Thallium (Tl)	EPA 279.2	5	ND	ND	ND	ND
Tin (Sn)	EPA 200.7	100	ND	ND	ND	ND
Zinc (Zn)	EPA 200.7	5	210	170	160	160

[a] Results expressed as μg/L.

TABLE A1-1. (CONTINUED) - PURGEABLE ORGANIC COMPOUNDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Benzene	EPA 524.2	1	ND	ND	ND	ND
Bromobenzene	EPA 524.2	1	ND	ND	ND	ND
Bromochloromethane	EPA 524.2	1	ND	ND	ND	ND
Bromodichloromethane	EPA 524.2	1	ND	ND	ND	ND
Bromoform	EPA 524.2	1	ND	ND	ND	ND
Bromomethane	EPA 524.2	1	ND	ND	ND	ND
Carbon tetrachloride	EPA 524.2	1	ND	ND	ND	ND
Chlorobenzene	EPA 524.2	1	2	ND	ND	ND
Chloroethane	EPA 524.2	1	ND	ND	ND	ND
Chloroform	EPA 524.2	1	10	1	ND	ND
Chloromethane	EPA 524.2	1	ND	ND	ND	ND
2-Chlorotoluene	EPA 524.2	1	ND	ND	ND	ND
4-Chlorotoluene	EPA 524.2	1	ND	ND	ND	ND
Dibromochloromethane	EPA 524.2	1	ND	ND	ND	ND
1,2-Dibromo-3-chloropropane	EPA 524.2	1	ND	ND	ND	ND
1,2-Dibromoethane	EPA 524.2	1	ND	ND	ND	ND
Dibromomethane	EPA 524.2	1	ND	ND	ND	ND
1,2-Dichlorobenzene	EPA 524.2	1	ND	ND	ND	ND
1,3-Dichlorobenzene	EPA 524.2	1	ND	ND	ND	ND
1,4-Dichlorobenzene	EPA 524.2	1	ND	ND	ND	ND
Dichlorodifluoromethane	EPA 524.2	1	ND	ND	ND	ND
1,1-Dichloroethane	EPA 524.2	1	ND	ND	ND	ND
1,2-Dichloroethane	EPA 524.2	1	ND	ND	ND	ND
1,1-Dichloroethene	EPA 524.2	1	1	ND	ND	ND
cis-1,2-Dichloroethene	EPA 524.2	1	430	27	ND	ND
trans-1,2-Dichloroethene	EPA 524.2	1	ND	ND	ND	ND
1,2-Dichloropropane	EPA 524.2	1	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-1. (CONTINUED) - PURGEABLE ORGANIC COMPOUNDS CON'T[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
1,3-Dichloropropane	EPA 524.2	1	ND	ND	ND	ND
2,2-Dichloropropane	EPA 524.2	1	ND	ND	ND	ND
1,1-Dichloropropene	EPA 524.2	1	ND	ND	ND	ND
cis-1,3-Dichloropropene	EPA 524.2	1	ND	ND	ND	ND
trans-1,3-Dichloropropene	EPA 524.2	1	ND	ND	ND	ND
Ethylbenzene	EPA 524.2	1	ND	ND	ND	ND
Methylene chloride	EPA 524.2	1	ND	ND	ND	ND
Styrene	EPA 524.2	1	ND	ND	ND	ND
1,1,1,2-Tetrachloroethane	EPA 524.2	1	3	ND	ND	ND
1,1,2,2-Tetrachloroethane	EPA 524.2	1	10000	1400	ND	7
Tetrachloroethene	EPA 524.2	1	90	6	ND	ND
Toluene	EPA 524.2	1	ND	ND	ND	ND
1,1,1-Trichloroethane	EPA 524.2	1	ND	ND	ND	ND
1,1,2-Trichloroethane	EPA 524.2	1	90	8	ND	1
Trichloroethene	EPA 524.2	1	1400	80	ND	ND
Trichlorofluoromethane	EPA 524.2	1	ND	ND	ND	ND
1,2,3-Trichloropropane	EPA 524.2	1	ND	ND	ND	ND
Vinyl chloride	EPA 524.2	1	2	ND	ND	ND
o-Xylene	EPA 524.2	1	ND	ND	ND	ND
m-Xylene	EPA 524.2	1	ND	ND	ND	ND
p-Xylene	EPA 524.2	1	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-1. (CONTINUED) - ORGANIC COMPOUNDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Acenaphthene	EPA 525	10	ND	ND	ND	ND
Acenaphthylene	EPA 525	10	ND	ND	ND	ND
Anthracene	EPA 525	10	ND	ND	ND	ND
Benzidine	EPA 525	20	ND	ND	ND	ND
Benzo[a]anthracene	EPA 525	10	ND	ND	ND	ND
Benzo[b]fluoranthene	EPA 525	10	ND	ND	ND	ND
Benzo[k]fluoranthene	EPA 525	10	ND	ND	ND	ND
Benzo[a]pyrene	EPA 525	10	ND	ND	ND	ND
Benzo[g,h,i]perylene	EPA 525	10	ND	ND	ND	ND
Butyl Benzyl Phthalate	EPA 525	10	ND	ND	ND	ND
Bis (2-chloroethyl) Ether	EPA 525	10	ND	ND	ND	ND
Bis (2-chloroethoxy) Methane	EPA 525	10	ND	ND	ND	ND
Bis (2-ethylhexyl) Phthalate	EPA 525	10	ND	ND	ND	ND
Bis (2-chloroisopropyl) Ether	EPA 525	10	ND	ND	ND	ND
4-Bromophenyl Phenyl Ether	EPA 525	10	ND	ND	ND	ND
2-Chloronapthalene	EPA 525	10	ND	ND	ND	ND
4-Chlorophenyl Phenyl Ether	EPA 525	10	ND	ND	ND	ND
Chrysene	EPA 525	10	ND	ND	ND	ND
Dibenzo (a,h) Anthracene	EPA 525	10	ND	ND	ND	ND
Di-n-Butylphthalate	EPA 525	10	ND	ND	ND	ND
1,3-Dichlorobenzene	EPA 525	10	ND	ND	ND	ND
1,2-Dichlorobenzene	EPA 525	10	ND	ND	ND	ND
1,4-Dichlorobenzene	EPA 525	10	ND	ND	ND	ND
3,3'-Dichlorobenzidine	EPA 525	20	ND	ND	ND	ND
1,2-Diphenylhydrazine	EPA 525	20	ND	ND	ND	ND
Diethyl Phthalate	EPA 525	10	ND	ND	ND	ND
Dimethyl Phthalate	EPA 525	10	ND	ND	ND	ND
2,4-Dinitrotoluene	EPA 525	10	ND	ND	ND	ND
2,6-Dinitrotoluene	EPA 525	10	ND	ND	ND	ND

[a] Results expressed as $\mu g/L$.

TABLE A1-1. (CONTINUED) - ORGANIC COMPOUNDS CON'T[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Di-n-octylphthalate	EPA 525	10	ND	ND	ND	ND
Fluoranthene	EPA 525	10	ND	ND	ND	ND
Fluorene	EPA 525	10	ND	ND	ND	ND
Hexachlorobenzene	EPA 525	10	ND	ND	ND	ND
Hexachlorobutadiene	EPA 525	10	ND	ND	ND	ND
Hexachloroethane	EPA 525	10	ND	ND	ND	ND
Hexachlorocyclopentadiene	EPA 525	10	ND	ND	ND	ND
Indeno[1,2,3,c,d]pyrene	EPA 525	10	ND	ND	ND	ND
Isophorone	EPA 525	10	ND	ND	ND	ND
Naphthalene	EPA 525	10	ND	ND	ND	ND
Nitrobenzene	EPA 525	10	ND	ND	ND	ND
N-Nitrosodi-n-propylamine	EPA 525	10	ND	ND	ND	ND
N-Nitrosodimethylamine	EPA 525	10	ND	ND	ND	ND
N-Nitrosodiphenylamine	EPA 525	10	ND	ND	ND	ND
Phenanthrene	EPA 525	10	ND	ND	ND	ND
Pyrene	EPA 525	10	ND	ND	ND	ND
1,2,4-Trichlorobenzene	EPA 525	10	ND	ND	ND	ND
4-Chloro-3-methylphenol	EPA 525	10	ND	ND	ND	ND
2-Chlorophenol	EPA 525	10	ND	ND	ND	ND
2,4-Dichlorophenol	EPA 525	10	ND	ND	ND	ND
2,4-Dimethylphenol	EPA 525	10	ND	ND	ND	ND
2,4-Dinitrophenol	EPA 525	50	ND	ND	ND	ND
2-Methyl-4,6-dinitrophenol	EPA 525	50	ND	ND	ND	ND
2-Nitrophenol	EPA 525	10	ND	ND	ND	ND
4-Nitrophenol	EPA 525	50	ND	ND	ND	ND
Pentachlorophenol	EPA 525	50	ND	ND	ND	ND
Phenol	EPA 525	10	ND	ND	ND	ND
2,4,6-Trichlorophenol	EPA 525	10	ND	ND	ND	ND

[a] Results expressed as $\mu g/L$.

TABLE A1-1. (CONTINUED) - CHLORINATED PESTICIDES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Aldrin	EPA 508	0.01	ND	ND	ND	ND
Chlordane-alpha	EPA 508	0.01	ND	ND	ND	ND
Chlordane-gamma	EPA 508	0.01	ND	ND	ND	ND
Chloroneb	EPA 508	0.01	ND	ND	ND	ND
Chlorobenzilate(a)	EPA 508	0.01	ND	ND	ND	ND
Chlorothalonil	EPA 508	0.01	ND	ND	ND	ND
DCPA	EPA 508	0.01	ND	ND	ND	ND
4,4'-DDD	EPA 508	0.01	ND	ND	ND	ND
4,4'-DDE	EPA 508	0.01	ND	ND	ND	ND
4,4'-DDT	EPA 508	0.01	ND	ND	ND	ND
Dieldrin	EPA 508	0.01	ND	ND	ND	ND
Endosulfan I	EPA 508	0.01	ND	ND	ND	ND
Endosulfan II	EPA 508	0.01	ND	ND	ND	ND
Endosulfan sulfate	EPA 508	0.01	ND	ND	ND	ND
Endrin	EPA 508	0.01	ND	ND	ND	ND
Endrin aldehyde	EPA 508	0.01	ND	ND	ND	ND
Etridiazole	EPA 508	0.01	ND	ND	ND	ND
HCH-alpha	EPA 508	0.01	ND	ND	ND	ND
HCH-beta	EPA 508	0.01	ND	ND	ND	ND
HCH-delta(a)	EPA 508	0.01	ND	ND	ND	ND
HCH-gamma(Lindane)	EPA 508	0.01	ND	ND	ND	ND
Heptachlor	EPA 508	0.01	ND	ND	ND	ND
Heptachlor epoxide	EPA 508	0.01	ND	ND	ND	ND
Hexachlorobenzene	EPA 508	0.01	ND	ND	ND	ND
Methoxychlor	EPA 508	0.01	ND	ND	ND	ND
cis-Permethrin	EPA 508	0.01	ND	ND	ND	ND
trans-Permethrin	EPA 508	0.01	ND	ND	ND	ND
Propachlor	EPA 508	0.01	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-1. (CONTINUED) - CHLORINATED PESTICIDES CON'T[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Trifluralin	EPA 508	0.01	ND	ND	ND	ND
Toxaphene	EPA 508	0.5	ND	ND	ND	ND
Chlordane	EPA 508	0.08	ND	ND	ND	ND
Aroclor 1016	EPA 508	0.1	ND	ND	ND	ND
Aroclor 1221	EPA 508	0.1	ND	ND	ND	ND
Aroclor 1232	EPA 508	0.1	ND	ND	ND	ND
Aroclor 1242	EPA 508	0.1	ND	ND	ND	ND
Aroclor 1248	EPA 508	0.1	ND	ND	ND	ND
Aroclor 1254	EPA 508	0.1	ND	ND	ND	ND
Aroclor 1260	EPA 508	0.1	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-1. (CONTINUED) - ORGANOPHOSPHORUS PESTICIDES AND CHLORINATED ACIDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Organophosphorus Pesticides:						
Malathion	EPA 614	0.2	ND	ND	ND	ND
Parathion	EPA 614	0.2	ND	ND	ND	ND
Dursban	EPA 614	0.2	ND	ND	ND	ND
Chlorinated Acids:						
Acifluorfen	EPA 515.1	0.05	ND	ND	ND	ND
Bentazon	EPA 515.1	0.05	ND	ND	ND	ND
Chloramben	EPA 515.1	0.05	ND	ND	ND	ND
2,4-D	EPA 515.1	0.05	ND	ND	ND	ND
Dalapon	EPA 515.1	0.05	ND	ND	ND	ND
2,4-DB	EPA 515.1	0.05	ND	ND	ND	ND
Dicamba	EPA 515.1	0.05	ND	ND	ND	ND
3,5-Dichlorobenzoic acid	EPA 515.1	0.05	ND	ND	ND	ND
Dichlorprop	EPA 515.1	0.05	ND	ND	ND	ND
Dinoseb	EPA 515.1	0.05	ND	ND	ND	ND
5-Hydroxydicamba	EPA 515.1	0.05	ND	ND	ND	ND
4-Nitrophenol	EPA 515.1	0.05	ND	ND	ND	ND
Pentachlorophenol (PCP)	EPA 515.1	0.05	ND	ND	ND	ND
Picloram	EPA 515.1	0.05	ND	ND	ND	ND
2,4,5-T	EPA 515.1	0.05	ND	ND	ND	ND
Methyl 2,4,5-TP	EPA 515.1	0.05	ND	ND	ND	ND
DCPA acid metabolites(a)	EPA 515.1	0.05	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-1. (CONTINUED) - NITROGEN AND PHOSPHOROUS-CONTAINING PESTICIDES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Alachlor	EPA 507	0.2	ND	ND	ND	ND
Ametryn	EPA 507	0.25	ND	ND	ND	ND
Atraton	EPA 507	0.2	ND	ND	ND	ND
Atrazine	EPA 507	0.25	ND	ND	ND	ND
Bromacil	EPA 507	0.2	ND	ND	ND	ND
Butachlor	EPA 507	0.2	ND	ND	ND	ND
Butylate	EPA 507	0.2	ND	ND	ND	ND
Carboxin	EPA 507	0.2	ND	ND	ND	ND
Chlorpropham	EPA 507	0.2	ND	ND	ND	ND
Cycloate	EPA 507	0.25	ND	ND	ND	ND
Diazinon(a)	EPA 507	0.2	ND	ND	ND	ND
Dichlorvos	EPA 507	0.2	ND	ND	ND	ND
Diphenamid	EPA 507	0.25	ND	ND	ND	ND
Disulfoton	EPA 507	0.25	ND	ND	ND	ND
Disulfoton sulfone	EPA 507	0.2	ND	ND	ND	ND
EPTC	EPA 507	0.25	ND	ND	ND	ND
Ethoprop	EPA 507	0.25	ND	ND	ND	ND
Fenamiphos	EPA 507	0.25	ND	ND	ND	ND
Fenarimol	EPA 507	0.2	ND	ND	ND	ND
Fluridone	EPA 507	1.0	ND	ND	ND	ND
Hexazinone	EPA 507	0.2	ND	ND	ND	ND
Merphos	EPA 507	0.25	ND	ND	ND	ND
Metolachlor	EPA 507	0.2	ND	ND	ND	ND
Metribuzin	EPA 507	0.2	ND	ND	ND	ND
Mevinphos	EPA 507	0.25	ND	ND	ND	ND
MGK 264	EPA 507	0.2	ND	ND	ND	ND
Molinate	EPA 507	0.2	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-1. (CONTINUED) - NITROGEN AND PHOSPHOROUS-CONTAINING PESTICIDES CON'T[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Napropamide	EPA 507	0.2	ND	ND	ND	ND
Norflurazon	EPA 507	0.2	ND	ND	ND	ND
Pebulate	EPA 507	0.2	ND	ND	ND	ND
Prometon	EPA 507	0.25	ND	ND	ND	ND
Prometryn	EPA 507	0.25	ND	ND	ND	ND
Pronamide(a)	EPA 507	0.2	ND	ND	ND	ND
Propazine	EPA 507	0.25	ND	ND	ND	ND
Simazine	EPA 507	0.25	ND	ND	ND	ND
Simetryn	EPA 507	0.2	ND	ND	ND	ND
Stirofos	EPA 507	0.2	ND	ND	ND	ND
Tebuthiuron	EPA 507	0.2	ND	ND	ND	ND
Terbacil	EPA 507	0.2	ND	ND	ND	ND
Terbufos(a)	EPA 507	0.2	ND	ND	ND	ND
Terbutryn	EPA 507	0.25	ND	ND	ND	ND
Triademefon	EPA 507	0.25	ND	ND	ND	ND
Tricyclazole	EPA 507	1.0	ND	ND	ND	ND
Vernolate	EPA 507	0.2	ND	ND	ND	ND

[a] Results expressed as $\mu g/L$.

TABLE A1-2. RESULTS OF THE APG-EA BEACH POINT GROUNDWATER AND DILUENT WATER COMPREHENSIVE CHEMICAL ANALYSES (TEST NO. 2) - GENERAL WATER QUALITY[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Total Alkalinity ($CaCO_3$)	EPA 310.1	1	ND	25	30	31
Ammonia (N)	SM 4500-NH_3 E	0.1	ND	ND	ND	ND
Chloride (Cl)	EPA 325.3	1	580	68	26	21
Total Cyanide (CN)	EPA 335.3	0.01	0.06	ND	ND	ND
Fluoride (F)	EPA 340.2	0.05	0.36	0.88	0.94	0.94
Hardness, ($CaCO_3$)	EPA 130.2	1	220	80	70	74
Nitrate (N)	EPA 353.2	0.1	2.5	3.2	2.3	2.5
Nitrite (N)	EPA 354.1	0.01	ND	ND	ND	ND
Total Phosphorus (P)	EPA 365.2	0.01	0.12	0.52	0.52	0.50
Total Suspended Solids	EPA 160.2	5	ND	ND	ND	ND
Specific Conductance	EPA 120.1	N/A	1760	356	213	200
Sulfate (SO_4)	EPA 375.4	1	62	17	12	12
Sulfide (S)	EPA 376.1	1	ND	ND	ND	ND
TOC	EPA 415.1	1	4	ND	ND	ND
Bromide (Br)	SM 4500-Br	[b]	1.5	0.44	0.44	0.37

[a] All results expressed as mg/L except for specific conductance which is expressed as μmohs/cm.

[b] The detection limit was 0.5 mg/L for the 100% groundwater sample and 0.1 mg/L for the 10% and 1% groundwater and dilution water samples.

TABLE A1-2. (CONTINUED) - METALS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Aluminum (Al)	EPA 200.7	0.1	1.1	ND	ND	ND
Antimony (Sb)	EPA 204.2	0.005	ND	ND	ND	ND
Arsenic (As)	EPA 206.2	0.005	ND	ND	ND	ND
Beryllium (Be)	EPA 200.7	0.005	ND	ND	ND	ND
Boron (B)	EPA 200.7	0.05	ND	ND	ND	ND
Cadmium (Cd)	EPA 213.2	0.0005	0.0008	0.0006	0.0007	0.0008
Calcium (Ca)	EPA 200.7	0.5	17	20	17	17
Chromium (Cr)	EPA 218.2	0.002	ND	ND	ND	ND
Cobalt (Co)	EPA 200.7	0.05	0.09	ND	ND	ND
Copper (Cu)	EPA 220.2	0.005	0.008	0.006	ND	ND
Iron (Fe)	EPA 200.7	0.1	5.4	0.5	0.1	ND
Lead (Pb)	EPA 239.2	0.005	ND	ND	ND	ND
Magnesium (Mg)	EPA 200.7	0.5	30	8.2	5.1	4.8
Manganese (Mn)	EPA 200.7	0.01	1.7	0.20	0.02	ND
Mercury (Hg)	EPA 245.1	0.0005	ND	ND	ND	ND
Molybdenum (Mo)	EPA 200.7	0.01	ND	ND	ND	ND
Nickel (Ni)	EPA 200.7	0.05	0.12	ND	ND	ND
Potassium (K)	EPA 200.7	0.5	2.5	2.0	1.8	1.7
Selenium (Se)	EPA 270.2	0.005	ND	ND	ND	ND
Silver (Ag)	EPA 272.2	0.001	ND	ND	ND	ND
Sodium (Na)	EPA 200.7	0.5	260	36	12	9.3
Thallium (Tl)	EPA 279.2	0.005	ND	ND	ND	ND
Tin (Sn)	EPA 282.2	0.005	ND	ND	ND	ND
Zinc (Zn)	EPA 200.7	0.05	0.24	0.16	0.13	0.12

[a] Results expressed as mg/L.

TABLE A1-2. (CONTINUED) - VOLATILES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Chloromethane	EPA 8240	10	ND	ND	ND	ND
Bromomethane	EPA 8240	10	ND	ND	ND	ND
Vinyl chloride	EPA 8240	10	ND	ND	ND	ND
Chloroethane	EPA 8240	10	ND	ND	ND	ND
Methylene chloride	EPA 8240	5	ND	ND	ND	ND
Acrolein	EPA 8240	100	ND	ND	ND	ND
Acrylonitrile	EPA 8240	100	ND	ND	ND	ND
Trichlorofluoromethane	EPA 8240	5	ND	ND	ND	ND
1,1-Dichloroethane	EPA 8240	5	ND	ND	ND	ND
1,2-Dichloroethene	EPA 8240	5	640	33	b	ND
Chloroform	EPA 8240	5	11	ND	ND	ND
1,2-Dichloroethane	EPA 8240	5	ND	ND	ND	ND
1,1,1-Trichloroethane	EPA 8240	5	ND	ND	ND	ND
Carbon tetrachloride	EPA 8240	5	ND	ND	ND	ND
Bromodichloromethane	EPA 8240	5	ND	ND	ND	ND
1,2-Dichloropropane	EPA 8240	5	ND	ND	ND	ND
1,3-Dichloropropene	EPA 8240	5	ND	ND	ND	ND
Dibromochloromethane	EPA 8240	5	ND	ND	ND	ND
1,1,2-Trichloroethane	EPA 8240	5	110	8	ND	ND
2-Chloroethylvinyl ether	EPA 8240	10	ND	ND	ND	ND
Bromoform	EPA 8240	5	ND	ND	ND	ND
Tetrachloroethene	EPA 8240	5	90	5	ND	ND
1,1,2,2-Tetrachloroethane	EPA 8240	5	17000	1300	160	9
Ethylbenzene	EPA 8240	5	ND	ND	ND	ND
1,1-Dichloroethene	EPA 8240	5	ND	ND	ND	ND
Trichloroethene	EPA 8240	5	1600	80	7	ND
Benzene	EPA 8240	5	ND	ND	ND	ND
Toluene	EPA 8240	5	ND	ND	ND	ND
Chlorobenzene	EPA 8240	5	b	ND	ND	ND

[a] Results expressed as μg/L.
[b] Detected below quantitation level.

TABLE A1-2. (CONTINUED) - BASE/NEUTRAL COMPOUNDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Acenaphthene	EPA 8270	10	ND	ND	ND	ND
2,4-Dinitrotoluene	EPA 8270	10	ND	ND	ND	ND
2,6-Dinitrotoluene	EPA 8270	10	ND	ND	ND	ND
Diethyl phthalate	EPA 8270	10	ND	ND	ND	ND
4-Chlorophenyl phenyl ether	EPA 8270	10	ND	ND	ND	ND
Fluorene	EPA 8270	10	ND	ND	ND	ND
N-Nitrosodiphenylamine	EPA 8270	10	ND	ND	ND	ND
1,2-Diphenylhydrazine	EPA 8270	10	ND	ND	ND	ND
4-Bromophenyl phenyl ether	EPA 8270	10	ND	ND	ND	ND
Hexachlorobenzene	EPA 8270	10	ND	ND	ND	ND
Phenanthrene	EPA 8270	10	ND	ND	ND	ND
Anthracene	EPA 8270	10	ND	ND	ND	ND
Di-n-butyl phthalate	EPA 8270	10	ND	ND	ND	ND
Fluoranthene	EPA 8270	10	ND	ND	ND	ND
Pyrene	EPA 8270	10	ND	ND	ND	ND
Benzidine	EPA 8270	50	ND	ND	ND	ND
Butylbenzyl phthalate	EPA 8270	10	ND	ND	ND	ND
3,3'-Dichlorobenzidine	EPA 8270	20	ND	ND	ND	ND
Benzo(a)anthracene	EPA 8270	10	ND	ND	ND	ND
bis(2-Ethylhexyl) phthalate	EPA 8270	10	ND	ND	ND	ND
Chrysene	EPA 8270	10	ND	ND	ND	ND
Di-n-octyl phthalate	EPA 8270	10	ND	ND	ND	ND
Benzo(b)fluoranthene	EPA 8270	10	ND	ND	ND	ND
Benzo(k)fluoranthene	EPA 8270	10	ND	ND	ND	ND
Benzo(a)pyrene	EPA 8270	10	ND	ND	ND	ND
Indeno(1,2,3-cd)pyrene	EPA 8270	10	ND	ND	ND	ND
Dibenz(a,h)anthracene	EPA 8270	10	ND	ND	ND	ND
Benzo(g,h,i)perylene	EPA 8270	10	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-2. (CONTINUED) - BASE/NEUTRAL COMPOUNDS CON'T[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
N-Nitrosodimethylamine	EPA 8270	10	ND	ND	ND	ND
bis(2-Chloroethyl) ether	EPA 8270	10	ND	ND	ND	ND
1,3-Dichlorobenzene	EPA 8270	10	ND	ND	ND	ND
1,4-Dichlorobenzene	EPA 8270	10	ND	ND	ND	ND
1,2-Dichlorobenzene	EPA 8270	10	ND	ND	ND	ND
bis(2-Chloroisopropyl) ether	EPA 8270	10	ND	ND	ND	ND
N-Nitrosodi-n-propylamine	EPA 8270	10	ND	ND	ND	ND
Hexachloroethane	EPA 8270	10	ND	ND	ND	ND
Nitrobenzene	EPA 8270	10	ND	ND	ND	ND
Isophorone	EPA 8270	10	ND	ND	ND	ND
bis(2-Chloroethoxy)methane	EPA 8270	10	ND	ND	ND	ND
1,2,4-Trichlorobenzene	EPA 8270	10	ND	ND	ND	ND
Naphthalene	EPA 8270	10	ND	ND	ND	ND
Hexachlorobutadiene	EPA 8270	10	ND	ND	ND	ND
Hexachlorocyclopentadiene	EPA 8270	10	ND	ND	ND	ND
2-Chloronaphthalene	EPA 8270	10	ND	ND	ND	ND
Dimethyl phthalate	EPA 8270	10	ND	ND	ND	ND
Acenaphthylene	EPA 8270	10	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-2. (CONTINUED) - PESTICIDES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
a-BHC	EPA 8080	0.05	ND	ND	ND	ND
b-BHC	EPA 8080	0.05	ND	ND	ND	ND
g-BHC	EPA 8080	0.05	ND	ND	ND	ND
d-BHC	EPA 8080	0.05	ND	ND	ND	ND
Heptachlor	EPA 8080	0.05	ND	ND	ND	ND
Aldrin	EPA 8080	0.05	ND	ND	ND	ND
Heptachlor epoxide	EPA 8080	0.05	ND	ND	ND	ND
a-Endosulfan	EPA 8080	0.1	ND	ND	ND	ND
4,4'-DDE	EPA 8080	0.1	ND	ND	ND	ND
Dieldrin	EPA 8080	0.1	ND	ND	ND	ND
Endrin	EPA 8080	0.1	ND	ND	ND	ND
b-Endosulfan	EPA 8080	0.3	ND	ND	ND	ND
Endrin aldehyde	EPA 8080	0.3	ND	ND	ND	ND
4,4'-DDD	EPA 8080	0.3	ND	ND	ND	ND
Endosulfan sulfate	EPA 8080	0.3	ND	ND	ND	ND
4,4'-DDT	EPA 8080	0.3	ND	ND	ND	ND
PCB-1242	EPA 8080	1	ND	ND	ND	ND
PCB-1254	EPA 8080	1	ND	ND	ND	ND
PCB-1221	EPA 8080	1	ND	ND	ND	ND
PCB-1232	EPA 8080	1	ND	ND	ND	ND
PCB-1248	EPA 8080	1	ND	ND	ND	ND
PCB-1260	EPA 8080	1	ND	ND	ND	ND
PCB-1016	EPA 8080	1	ND	ND	ND	ND
Chlordane	EPA 8080	1	ND	ND	ND	ND
Toxaphene	EPA 8080	3	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-2. (CONTINUED) - HERBICIDES AND ORGANOPHOSPHORUS PESTICIDES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Herbicides:						
2,4-D	EPA 8150	0.5	ND	ND	ND	ND
2,4,5-TP (Silvex)	EPA 8150	0.2	ND	ND	ND	ND
Organophosphorus Pesticides:						
Malathion	EPA 8080	0.5	ND	ND	ND	ND
Methyl azinphos	EPA 8080	3	ND	ND	ND	ND
Coumaphos	EPA 8080	2	ND	ND	ND	ND
Chlorpyrifos	EPA 8080	0.1	ND	ND	ND	ND
Diazinon	EPA 8080	0.5	ND	ND	ND	ND
Fenthion	EPA 8080	50	ND	ND	ND	ND
Naled	EPA 8080	5	ND	ND	ND	ND
Methyl parathion	EPA 8080	0.5	ND	ND	ND	ND
Parathion	EPA 8080	0.5	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-2. (CONTINUED) - ACID COMPOUNDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Phenol	EPA 8270	10	ND	ND	ND	ND
2-Chlorophenol	EPA 8270	10	ND	ND	ND	ND
2-Nitrophenol	EPA 8270	10	ND	ND	ND	ND
2,4-Dimethylphenol	EPA 8270	10	ND	ND	ND	ND
2,4-Dichlorophenol	EPA 8270	10	ND	ND	ND	ND
4-Chloro-3-methylphenol	EPA 8270	20	ND	ND	ND	ND
2,4,6-Trichlorophenol	EPA 8270	10	ND	ND	ND	ND
2,4-Dinitrophenol	EPA 8270	50	ND	ND	ND	ND
4-Nitrophenol	EPA 8270	50	ND	ND	ND	ND
4,6-Dinitro-2-methylphenol	EPA 8270	50	ND	ND	ND	ND
Pentachlorophenol	EPA 8270	50	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-3. RESULTS OF THE APG-EA BEACH POINT GROUNDWATER AND DILUENT WATER COMPREHENSIVE CHEMICAL ANALYSES (TEST NO. 3) - GENERAL WATER QUALITY[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Total Alkalinity ($CaCO_3$)	EPA 310.1	1	ND	20	22	25
Ammonia (N)	SM 4500-NH_3 E	0.1	ND	ND	ND	ND
Chloride (Cl)	EPA 325.3	[b]	580	78	27	21
Total Cyanide (CN)	EPA 335.3	0.01	ND	ND	ND	ND
Fluoride (F)	EPA 340.2	0.05	0.24	0.88	0.98	0.94
Hardness, ($CaCO_3$)	EPA 130.2	[c]	300	92	76	81
Nitrate (N)	EPA 353.2	0.1	ND	1.7	1.8	1.8
Nitrite (N)	EPA 354.1	0.01	ND	ND	ND	ND
Total Phosphorus (P)	EPA 365.2	0.01	0.01	0.39	0.44	0.43
Total Suspended Solids	EPA 160.2	5	ND	ND	ND	ND
Specific Conductance	EPA 120.1	N/A	1680	376	212	196
Sulfate (SO_4)	EPA 375.4	1	75	21	16	17
Sulfide (S)	EPA 376.1	1	ND	ND	ND	ND
TOC	EPA 415.1	1	2	ND	ND	ND
Bromide (Br)	SM 4500-Br	0.1	0.83	0.26	ND	ND

[a] All results expressed as mg/L except for specific conductance which is expressed as μmohs/cm.

[b] The detection limit was 10 mg/L for the 100% groundwater sample and 1 mg/L for the 10% and 1% groundwater and dilution water samples.

[c] The detection limit was 5 mg/L for the 100% groundwater sample and 1 mg/L for the 10% and 1% groundwater and dilution water samples.

TABLE A1-3. (CONTINUED) - METALS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Aluminum (Al)	EPA 200.7	0.1	1.3	0.1	ND	ND
Antimony (Sb)	EPA 204.2	0.005	ND	ND	ND	ND
Arsenic (As)	EPA 206.2	0.005	ND	ND	ND	ND
Beryllium (Be)	EPA 200.7	0.005	ND	ND	ND	ND
Boron (B)	EPA 200.7	0.05	0.05	ND	ND	ND
Cadmium (Cd)	EPA 213.2	0.0005	0.0014	ND	ND	ND
Calcium (Ca)	EPA 200.7	0.5	19	16	16	16
Chromium (Cr)	EPA 218.2	0.002	ND	ND	ND	ND
Cobalt (Co)	EPA 200.7	0.02	0.10	ND	ND	ND
Copper (Cu)	EPA 220.2	0.005	0.008	ND	ND	ND
Iron (Fe)	EPA 200.7	0.1	7.5	0.62	0.13	ND
Lead (Pb)	EPA 239.2	0.005	ND	ND	ND	ND
Magnesium (Mg)	EPA 200.7	0.5	37	7.7	4.1	4.2
Manganese (Mn)	EPA 200.7	0.01	2.1	0.22	0.02	ND
Mercury (Hg)	EPA 245.1	0.0005	ND	ND	ND	ND
Molybdenum (Mo)	EPA 200.7	0.02	ND	ND	ND	ND
Nickel (Ni)	EPA 200.7	0.02	0.16	0.02	0.06	0.03
Potassium (K)	EPA 200.7	0.5	3.1	2.5	1.7	2.0
Selenium (Se)	EPA 270.2	0.005	ND	ND	ND	ND
Silver (Ag)	EPA 272.2	0.001	ND	ND	ND	ND
Sodium (Na)	EPA 200.7	0.5	300	40	12	10
Thallium (Tl)	EPA 279.2	0.005	ND	ND	ND	ND
Tin (Sn)	EPA 282.2	0.005	ND	ND	ND	ND
Zinc (Zn)	EPA 200.7	0.05	0.30	0.44	0.45	0.47

[a] Results expressed as mg/L.

TABLE A1-3. (CONTINUED) - VOLATILES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Chloromethane	EPA 8021	1	ND	ND	ND	ND
Bromomethane	EPA 8021	1	ND	ND	ND	ND
Vinyl chloride	EPA 8021	1	ND	ND	ND	ND
Chloroethane	EPA 8021	1	ND	ND	ND	ND
Methylene chloride	EPA 8021	1	ND	ND	ND	ND
Trichlorofluoromethane	EPA 8021	1	ND	ND	ND	ND
1,1-Dichloroethane	EPA 8021	1	ND	ND	ND	ND
1,2-Dichloroethene	EPA 8021	1	140	6	[b]	ND
Chloroform	EPA 8021	1	12	5	5	7
1,2-Dichloroethane	EPA 8021	1	ND	ND	ND	ND
1,1,1-Trichloroethane	EPA 8021	1	ND	ND	ND	ND
Carbon tetrachloride	EPA 8021	1	ND	ND	ND	ND
Bromodichloromethane	EPA 8021	1	ND	ND	ND	1
1,2-Dichloropropane	EPA 8021	1	ND	ND	ND	ND
1,3-Dichloropropene	EPA 8021	1	ND	ND	ND	ND
Dibromochloromethane	EPA 8021	1	ND	ND	ND	ND
1,1,2-Trichloroethane	EPA 8021	1	37	5	ND	ND
2-Chloroethylvinyl ether	EPA 8021	5	ND	ND	ND	ND
Bromoform	EPA 8021	2	ND	ND	ND	ND
Tetrachloroethene	EPA 8021	1	30	4	ND	ND
1,1,2,2-Tetrachloroethane	EPA 8021	[c]	10000	900	100	8
Ethylbenzene	EPA 8021	1	ND	ND	ND	ND
1,1-Dichloroethene	EPA 8021	1	ND	ND	ND	ND
Trichloroethene	EPA 8021	[d]	450	50	5	ND
Benzene	EPA 8021	1	ND	ND	ND	ND
Toluene	EPA 8021	1	ND	ND	ND	ND
Chlorobenzene	EPA 8021	1	2	ND	ND	ND

[a] Results expressed as μg/L.
[b] Detected below quantitation level.
[c] Detection limits were as follows: 1000 μg/L for the 100% groundwater sample, 100 μg/L for the 10% groundwater sample, 10 μg/L for the 1% groundwater sample, and 1 μg/L for the diluent water sample.
[d] Detection limits were as follows: 20 μg/L for the 100% groundwater sample, 10 μg/L for the 10% groundwater sample, and 1 μg/L for the 1% groundwater and diluent water samples.

TABLE A1-3. (CONTINUED) - BASE/NEUTRAL COMPOUNDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Acenaphthene	EPA 8270	10	ND	ND	ND	ND
2,4-Dinitrotoluene	EPA 8270	10	ND	ND	ND	ND
2,6-Dinitrotoluene	EPA 8270	10	ND	ND	ND	ND
Diethyl phthalate	EPA 8270	10	ND	ND	ND	ND
4-Chlorophenyl phenyl ether	EPA 8270	10	ND	ND	ND	ND
Fluorene	EPA 8270	10	ND	ND	ND	ND
N-Nitrosodiphenylamine	EPA 8270	10	ND	ND	ND	ND
1,2-Diphenylhydrazine	EPA 8270	10	ND	ND	ND	ND
4-Bromophenyl phenyl ether	EPA 8270	10	ND	ND	ND	ND
Hexachlorobenzene	EPA 8270	10	ND	ND	ND	ND
Phenanthrene	EPA 8270	10	ND	ND	ND	ND
Anthracene	EPA 8270	10	ND	ND	ND	ND
Di-n-butyl phthalate	EPA 8270	10	ND	ND	ND	ND
Fluoranthene	EPA 8270	10	ND	ND	ND	ND
Pyrene	EPA 8270	10	ND	ND	ND	ND
Benzidine	EPA 8270	50	ND	ND	ND	ND
Butylbenzyl phthalate	EPA 8270	10	ND	ND	ND	ND
3,3'-Dichlorobenzidine	EPA 8270	20	ND	ND	ND	ND
Benzo(a)anthracene	EPA 8270	10	ND	ND	ND	ND
bis(2-Ethylhexyl) phthalate	EPA 8270	10	ND	ND	ND	ND
Chrysene	EPA 8270	10	ND	ND	ND	ND
Di-n-octyl phthalate	EPA 8270	10	ND	ND	ND	ND
Benzo(b)fluoranthene	EPA 8270	10	ND	ND	ND	ND
Benzo(k)fluoranthene	EPA 8270	10	ND	ND	ND	ND
Benzo(a)pyrene	EPA 8270	10	ND	ND	ND	ND
Indeno(1,2,3-cd)pyrene	EPA 8270	10	ND	ND	ND	ND
Dibenz(a,h)anthracene	EPA 8270	10	ND	ND	ND	ND
Benzo(g,h,i)perylene	EPA 8270	10	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-3. (CONTINUED) - BASE/NEUTRAL COMPOUNDS CON'T[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
N-Nitrosodimethylamine	EPA 8270	10	ND	ND	ND	ND
bis(2-Chloroethyl) ether	EPA 8270	10	ND	ND	ND	ND
1,3-Dichlorobenzene	EPA 8270	10	ND	ND	ND	ND
1,4-Dichlorobenzene	EPA 8270	10	ND	ND	ND	ND
1,2-Dichlorobenzene	EPA 8270	10	ND	ND	ND	ND
bis(2-Chloroisopropyl) ether	EPA 8270	10	ND	ND	ND	ND
N-Nitrosodi-n-propylamine	EPA 8270	10	ND	ND	ND	ND
Hexachloroethane	EPA 8270	10	ND	ND	ND	ND
Nitrobenzene	EPA 8270	10	ND	ND	ND	ND
Isophorone	EPA 8270	10	ND	ND	ND	ND
bis(2-Chloroethoxy)methane	EPA 8270	10	ND	ND	ND	ND
1,2,4-Trichlorobenzene	EPA 8270	10	ND	ND	ND	ND
Naphthalene	EPA 8270	10	ND	ND	ND	ND
Hexachlorobutadiene	EPA 8270	10	ND	ND	ND	ND
Hexachlorocyclopentadiene	EPA 8270	10	ND	ND	ND	ND
2-Chloronaphthalene	EPA 8270	10	ND	ND	ND	ND
Dimethyl phthalate	EPA 8270	10	ND	ND	ND	ND
Acenaphthylene	EPA 8270	10	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-3. (CONTINUED) - PESTICIDES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
a-BHC	EPA 8080	0.05	ND	ND	ND	ND
b-BHC	EPA 8080	0.05	ND	ND	ND	ND
g-BHC	EPA 8080	0.05	ND	ND	ND	ND
d-BHC	EPA 8080	0.05	ND	ND	ND	ND
Heptachlor	EPA 8080	0.05	ND	ND	ND	ND
Aldrin	EPA 8080	0.05	ND	ND	ND	ND
Heptachlor epoxide	EPA 8080	0.05	ND	ND	ND	ND
a-Endosulfan	EPA 8080	0.1	ND	ND	ND	ND
4,4'-DDE	EPA 8080	0.1	ND	ND	ND	ND
Dieldrin	EPA 8080	0.1	ND	ND	ND	ND
Endrin	EPA 8080	0.1	ND	ND	ND	ND
b-Endosulfan	EPA 8080	0.3	ND	ND	ND	ND
Endrin aldehyde	EPA 8080	0.3	ND	ND	ND	ND
4,4'-DDD	EPA 8080	0.3	ND	ND	ND	ND
Endosulfan sulfate	EPA 8080	0.3	ND	ND	ND	ND
4,4'-DDT	EPA 8080	0.3	ND	ND	ND	ND
PCB-1242	EPA 8080	1	ND	ND	ND	ND
PCB-1254	EPA 8080	1	ND	ND	ND	ND
PCB-1221	EPA 8080	1	ND	ND	ND	ND
PCB-1232	EPA 8080	1	ND	ND	ND	ND
PCB-1248	EPA 8080	1	ND	ND	ND	ND
PCB-1260	EPA 8080	1	ND	ND	ND	ND
PCB-1016	EPA 8080	1	ND	ND	ND	ND
Chlordane	EPA 8080	1	ND	ND	ND	ND
Toxaphene	EPA 8080	3	ND	ND	ND	ND

[a] Results expressed as $\mu g/L$.

TABLE A1-3. (CONTINUED) - HERBICIDES AND ORGANOPHOSPHORUS PESTICIDES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Herbicides:						
2,4-D	EPA 8150	0.5	ND	ND	ND	ND
2,4,5-TP (Silvex)	EPA 8150	0.2	ND	ND	ND	ND
Organophosphorus Pesticides:						
Malathion	EPA 8080	0.5	ND	ND	ND	ND
Methyl azinphos	EPA 8080	3	ND	ND	ND	ND
Coumaphos	EPA 8080	2	ND	ND	ND	ND
Chlorpyrifos	EPA 8080	0.1	ND	ND	ND	ND
Diazinon	EPA 8080	0.5	ND	ND	ND	ND
Fenthion	EPA 8080	50	ND	ND	ND	ND
Naled	EPA 8080	5	ND	ND	ND	ND
Methyl parathion	EPA 8080	0.5	ND	ND	ND	ND
Parathion	EPA 8080	0.5	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-3. (CONTINUED) - ACID COMPOUNDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Phenol	EPA 8270	10	ND	ND	ND	ND
2-Chlorophenol	EPA 8270	10	ND	ND	ND	ND
2-Nitrophenol	EPA 8270	10	ND	ND	ND	ND
2,4-Dimethylphenol	EPA 8270	10	ND	ND	ND	ND
2,4-Dichlorophenol	EPA 8270	10	ND	ND	ND	ND
4-Chloro-3-methylphenol	EPA 8270	20	ND	ND	ND	ND
2,4,6-Trichlorophenol	EPA 8270	10	ND	ND	ND	ND
2,4-Dinitrophenol	EPA 8270	50	ND	ND	ND	ND
4-Nitrophenol	EPA 8270	50	ND	ND	ND	ND
4,6-Dinitro-2-methylphenol	EPA 8270	50	ND	ND	ND	ND
Pentachlorophenol	EPA 8270	50	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-4. RESULTS OF THE APG-EA BEACH POINT GROUNDWATER AND DILUENT WATER COMPREHENSIVE CHEMICAL ANALYSES (TEST NO. 4) - GENERAL WATER QUALITY[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Total Alkalinity ($CaCO_3$)	EPA 310.1	1	ND	18	23	21
Ammonia (N)	SM 4500-NH_3 E	0.1	ND	ND	ND	ND
Chloride (Cl)	EPA 325.3	[b]	570	72	25	19
Total Cyanide (CN)	EPA 335.3	0.01	ND	ND	ND	ND
Fluoride (F)	EPA 340.2	0.1	0.2	0.6	0.7	0.7
Hardness, ($CaCO_3$)	EPA 130.2	[c]	230	89	68	76
Nitrate (N)	EPA 353.2	0.1	ND	1.9	2.1	2.2
Nitrite (N)	EPA 354.1	0.01	ND	ND	ND	ND
Total Phosphorus (P)	EPA 365.2	0.01	ND	0.32	0.36	0.37
Total Suspended Solids	EPA 160.2	5	ND	ND	ND	ND
Specific Conductance	EPA 120.1	N/A	2020	388	210	180
Sulfate (SO_4)	EPA 375.4	[c]	82	29	22	19
Sulfide (S)	EPA 376.1	1	ND	ND	ND	ND
TOC	EPA 415.1	1	3	ND	ND	ND
Bromide (Br)	SM 4500-Br	0.1	0.4	0.3	0.1	ND

[a] All results expressed as mg/L except for specific conductance which is expressed as μmohs/cm.

[b] The detection limit was 10 mg/L for the 100% groundwater sample and 1 mg/L for the 10% and 1% groundwater and dilution water samples.

[c] The detection limit was 2 mg/L for the 100% groundwater sample and 1 mg/L for the 10% and 1% groundwater and dilution water samples.

TABLE A1-4. (CONTINUED) - METALS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Aluminum (Al)	EPA 200.7	0.1	1.5	0.3	ND	ND
Antimony (Sb)	EPA 204.2	0.005	ND	ND	ND	ND
Arsenic (As)	EPA 206.2	0.005	0.007	ND	ND	ND
Beryllium (Be)	EPA 200.7	0.005	ND	ND	ND	ND
Boron (B)	EPA 200.7	0.05	0.07	ND	ND	ND
Cadmium (Cd)	EPA 213.2	0.0005	0.0009	ND	ND	ND
Calcium (Ca)	EPA 200.7	0.5	21	19	17	17
Chromium (Cr)	EPA 218.2	0.002	ND	ND	ND	ND
Cobalt (Co)	EPA 200.7	0.05	0.08	ND	ND	ND
Copper (Cu)	EPA 220.2	0.005	0.006	ND	ND	ND
Iron (Fe)	EPA 200.7	0.1	7.7	0.8	0.3	ND
Lead (Pb)	EPA 239.2	0.005	ND	ND	ND	ND
Magnesium (Mg)	EPA 200.7	0.5	43	9.9	6.1	5.5
Manganese (Mn)	EPA 200.7	0.01	2.1	0.22	0.03	ND
Mercury (Hg)	EPA 245.1	0.0005	ND	ND	ND	ND
Molybdenum (Mo)	EPA 200.7	0.05	ND	ND	ND	ND
Nickel (Ni)	EPA 200.7	0.05	0.18	ND	ND	ND
Potassium (K)	EPA 200.7	0.5	3.9	2.9	2.9	2.7
Selenium (Se)	EPA 270.2	0.005	ND	ND	ND	ND
Silver (Ag)	EPA 272.2	0.001	ND	ND	ND	ND
Sodium (Na)	EPA 200.7	0.5	350	41	13	9.1
Thallium (Tl)	EPA 279.2	0.005	ND	ND	ND	ND
Tin (Sn)	EPA 282.2	0.005	ND	ND	ND	ND
Zinc (Zn)	EPA 200.7	0.05	0.31	0.62	0.66	0.60

[a] Results expressed as mg/L.

TABLE A1-4. (CONTINUED) - VOLATILES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Chloromethane	EPA 8021	2	ND	ND	ND	ND
Bromomethane	EPA 8021	5	ND	ND	ND	ND
Vinyl chloride	EPA 8021	2	ND	ND	ND	ND
Chloroethane	EPA 8021	1	ND	ND	ND	ND
Methylene chloride	EPA 8021	1	ND	ND	ND	ND
Trichlorofluoromethane	EPA 8021	1	ND	ND	ND	ND
1,1-Dichloroethane	EPA 8021	1	ND	ND	ND	ND
1,2-Dichloroethene	EPA 8021	1	110	4	ND	ND
Chloroform	EPA 8021	1	12	4	3	3
1,2-Dichloroethane	EPA 8021	1	ND	ND	ND	ND
1,1,1-Trichloroethane	EPA 8021	1	ND	ND	ND	ND
Carbon tetrachloride	EPA 8021	1	ND	ND	ND	ND
Bromodichloromethane	EPA 8021	1	ND	ND	ND	ND
1,2-Dichloropropane	EPA 8021	1	ND	ND	ND	ND
1,3-Dichloropropene	EPA 8021	1	ND	ND	ND	ND
Dibromochloromethane	EPA 8021	1	ND	ND	ND	ND
1,1,2-Trichloroethane	EPA 8021	1	70	4	ND	ND
2-Chloroethylvinyl ether	EPA 8021	5	ND	ND	ND	ND
Bromoform	EPA 8021	5	ND	ND	ND	ND
Tetrachloroethene	EPA 8021	1	80	3	ND	ND
1,1,2,2-Tetrachloroethane	EPA 8021	[b]	9000	700	91	4
Ethylbenzene	EPA 8021	1	ND	ND	ND	ND
1,1-Dichloroethene	EPA 8021	1	ND	ND	ND	ND
Trichloroethene	EPA 8021	1	780	40	4	ND
Benzene	EPA 8021	1	ND	ND	ND	ND
Toluene	EPA 8021	1	ND	ND	ND	ND
Chlorobenzene	EPA 8021	1	ND	ND	ND	ND

[a] Results expressed as μg/L.
[b] The detection limit was 10 μg/L for the 100% groundwater sample and 1 μg/L for the 10% and 1% groundwater and diluent water samples.

TABLE A1-4. (CONTINUED) - BASE/NEUTRAL COMPOUNDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Acenaphthene	EPA 8270A	10	ND	ND	ND	ND
2,4-Dinitrotoluene	EPA 8270A	10	ND	ND	ND	ND
2,6-Dinitrotoluene	EPA 8270A	10	ND	ND	ND	ND
Diethyl phthalate	EPA 8270A	10	ND	ND	ND	ND
4-Chlorophenyl phenyl ether	EPA 8270A	10	ND	ND	ND	ND
Fluorene	EPA 8270A	10	ND	ND	ND	ND
N-Nitrosodiphenylamine	EPA 8270A	10	ND	ND	ND	ND
1,2-Diphenylhydrazine	EPA 8270A	10	ND	ND	ND	ND
4-Bromophenyl phenyl ether	EPA 8270A	10	ND	ND	ND	ND
Hexachlorobenzene	EPA 8270A	10	ND	ND	ND	ND
Phenanthrene	EPA 8270A	10	ND	ND	ND	ND
Anthracene	EPA 8270A	10	ND	ND	ND	ND
Di-n-butyl phthalate	EPA 8270A	10	ND	ND	ND	ND
Fluoranthene	EPA 8270A	10	ND	ND	ND	ND
Pyrene	EPA 8270A	10	ND	ND	ND	ND
Benzidine	EPA 8270A	50	ND	ND	ND	ND
Butylbenzyl phthalate	EPA 8270A	10	ND	ND	ND	ND
3,3'-Dichlorobenzidine	EPA 8270A	20	ND	ND	ND	ND
Benzo(a)anthracene	EPA 8270A	10	ND	ND	ND	ND
bis(2-Ethylhexyl) phthalate	EPA 8270A	10	ND	ND	ND	ND
Chrysene	EPA 8270A	10	ND	ND	ND	ND
Di-n-octyl phthalate	EPA 8270A	10	ND	ND	ND	ND
Benzo(b)fluoranthene	EPA 8270A	10	ND	ND	ND	ND
Benzo(k)fluoranthene	EPA 8270A	10	ND	ND	ND	ND
Benzo(a)pyrene	EPA 8270A	10	ND	ND	ND	ND
Indeno(1,2,3-cd)pyrene	EPA 8270A	10	ND	ND	ND	ND
Dibenz(a,h)anthracene	EPA 8270A	10	ND	ND	ND	ND
Benzo(g,h,i)perylene	EPA 8270A	10	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-4. (CONTINUED) - BASE/NEUTRAL COMPOUNDS CON'T[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
N-Nitrosodimethylamine	EPA 8270A	10	ND	ND	ND	ND
bis(2-Chloroethyl) ether	EPA 8270A	10	ND	ND	ND	ND
1,3-Dichlorobenzene	EPA 8270A	10	ND	ND	ND	ND
1,4-Dichlorobenzene	EPA 8270A	10	ND	ND	ND	ND
1,2-Dichlorobenzene	EPA 8270A	10	ND	ND	ND	ND
bis(2-Chloroisopropyl) ether	EPA 8270A	10	ND	ND	ND	ND
N-Nitrosodi-n-propylamine	EPA 8270A	10	ND	ND	ND	ND
Hexachloroethane	EPA 8270A	10	ND	ND	ND	ND
Nitrobenzene	EPA 8270A	10	ND	ND	ND	ND
Isophorone	EPA 8270A	10	ND	ND	ND	ND
bis(2-Chloroethoxy)methane	EPA 8270A	10	ND	ND	ND	ND
1,2,4-Trichlorobenzene	EPA 8270A	10	ND	ND	ND	ND
Naphthalene	EPA 8270A	10	ND	ND	ND	ND
Hexachlorobutadiene	EPA 8270A	10	ND	ND	ND	ND
Hexachlorocyclopentadiene	EPA 8270A	10	ND	ND	ND	ND
2-Chloronaphthalene	EPA 8270A	10	ND	ND	ND	ND
Dimethyl phthalate	EPA 8270A	10	ND	ND	ND	ND
Acenaphthylene	EPA 8270A	10	ND	ND	ND	ND

[a] Results expressed as $\mu g/L$.

TABLE A1-4. (CONTINUED) - PESTICIDES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
a-BHC	EPA 8080	0.05	ND	ND	ND	ND
b-BHC	EPA 8080	0.05	ND	ND	ND	ND
g-BHC	EPA 8080	0.05	ND	ND	ND	ND
d-BHC	EPA 8080	0.05	ND	ND	ND	ND
Heptachlor	EPA 8080	0.05	ND	ND	ND	ND
Aldrin	EPA 8080	0.05	ND	ND	ND	ND
Heptachlor epoxide	EPA 8080	0.05	ND	ND	ND	ND
a-Endosulfan	EPA 8080	0.1	ND	ND	ND	ND
4,4'-DDE	EPA 8080	0.1	ND	ND	ND	ND
Dieldrin	EPA 8080	0.1	ND	ND	ND	ND
Endrin	EPA 8080	0.1	ND	ND	ND	ND
b-Endosulfan	EPA 8080	0.3	ND	ND	ND	ND
Endrin aldehyde	EPA 8080	0.3	ND	ND	ND	ND
4,4'-DDD	EPA 8080	0.3	ND	ND	ND	ND
Endosulfan sulfate	EPA 8080	0.3	ND	ND	ND	ND
4,4'-DDT	EPA 8080	0.3	ND	ND	ND	ND
PCB-1242	EPA 8080	1	ND	ND	ND	ND
PCB-1254	EPA 8080	1	ND	ND	ND	ND
PCB-1221	EPA 8080	1	ND	ND	ND	ND
PCB-1232	EPA 8080	1	ND	ND	ND	ND
PCB-1248	EPA 8080	1	ND	ND	ND	ND
PCB-1260	EPA 8080	1	ND	ND	ND	ND
PCB-1016	EPA 8080	1	ND	ND	ND	ND
Chlordane	EPA 8080	1	ND	ND	ND	ND
Toxaphene	EPA 8080	3	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-4. (CONTINUED) - HERBICIDES AND ORGANOPHOSPHORUS PESTICIDES[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Herbicides:						
2,4-D	EPA 8150A	0.5	ND	ND	ND	ND
2,4,5-TP (Silvex)	EPA 8150A	0.2	ND	ND	ND	ND
Organophosphorus Pesticides:						
Malathion	EPA 8080	0.5	ND	ND	ND	ND
Methyl azinphos	EPA 8080	3	ND	ND	ND	ND
Coumaphos	EPA 8080	2	ND	ND	ND	ND
Chlorpyrifos	EPA 8080	0.1	ND	ND	ND	ND
Diazinon	EPA 8080	0.5	ND	ND	ND	ND
Fenthion	EPA 8080	50	ND	ND	ND	ND
Naled	EPA 8080	5	ND	ND	ND	ND
Methyl parathion	EPA 8080	0.5	ND	ND	ND	ND
Parathion	EPA 8080	0.5	ND	ND	ND	ND

[a] Results expressed as μg/L.

TABLE A1-4. (CONTINUED) - ACID COMPOUNDS[a]

Parameter	Method	Detection Limits	100% G.W.	10% G.W.	1% G.W.	Diluent H_2O
Phenol	EPA 8270A	10	ND	ND	ND	ND
2-Chlorophenol	EPA 8270A	10	ND	ND	ND	ND
2-Nitrophenol	EPA 8270A	10	ND	ND	ND	ND
2,4-Dimethylphenol	EPA 8270A	10	ND	ND	ND	ND
2,4-Dichlorophenol	EPA 8270A	10	ND	ND	ND	ND
4-Chloro-3-methylphenol	EPA 8270A	20	ND	ND	ND	ND
2,4,6-Trichlorophenol	EPA 8270A	10	ND	ND	ND	ND
2,4-Dinitrophenol	EPA 8270A	50	ND	ND	ND	ND
4-Nitrophenol	EPA 8270A	50	ND	ND	ND	ND
4,6-Dinitro-2-methylphenol	EPA 8270A	50	ND	ND	ND	ND
Pentachlorophenol	EPA 8270A	50	ND	ND	ND	ND

[a] Results expressed as μg/L.

APPENDIX 2

SCREENING-LEVEL ASSESSMENT OF THE NEAR- AND FAR-FIELD DILUTION OF THE SURFICIAL GROUNDWATER DISCHARGE FROM BEACH POINT INTO THE BUSH RIVER

FINAL REPORT - JOB 1437

SCREENING-LEVEL ASSESSMENT OF THE NEAR-FIELD DILUTION OF GROUNDWATER DISCHARGES FROM THE BEACH POINT SITE

Prepared for:
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JULY, 1994

EXECUTIVE SUMMARY

This study was initiated to assess the potential impact to the Bush River due to contaminated groundwater emanating from the Army Proving Ground site in Harford County, Maryland. Towards this objective, an EPA-approved near-field model was adapted to the receiving-waters adjacent to the APG site. The model approach and input was developed to provide a conservative assessment of receiving water impacts. This analysis suggests a near-field dilution level of approximately 42:1 for the application of acute criteria and a near-field dilution level of approximately 168:1 for the application of chronic criteria. Thus, contaminants introduced into Bush River receiving-waters at a concentration of 42 ppb will be diluted locally to a concentration of approximately 1 ppb or less. By applying these dilution factors to the groundwater quality data from the APG site, near-field concentrations within the Bush River can be projected. These results indicate that Tables 8 and 9, none of the projected receiving water concentrations would exceed Maryland's current acute or chronic criteria.

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1. INTRODUCTION

1.1 Background

The U.S Army Aberdeen Proving Ground (APG) is located between the Bush River and the Upper Chesapeake Bay in Harford County, Maryland. Currently, the U.S. Army Biomedical Research and Development Laboratory (USABRDL) is conducting biomonitoring evaluation studies of groundwater contamination at the APG site. To complement such studies, the USABRDL requires information regarding potential impacts of such groundwater contamination on Bush River water quality. Such impacts depend upon the ability of these receiving waters to mix and dilute contaminants introduced by groundwater sources. Accordingly, Najarian Associates, Inc. was contracted by the USABRDL to conduct preliminary model studies of groundwater-plume dilution in the Bush River.

The initial dilution of groundwater-discharge plumes depends, in part, upon the physical characteristics of both the source and the ambient receiving waters. These characteristics include the quantity/quality of the discharge, the geometrical configuration of the source, the ambient current speed, the ambient density stratification, and the water depth. These latter three factors control the near-field mixing of discharged constituents with surrounding receiving-waters, primarily through the mechanisms of plume buoyancy and initial discharge momentum. Typically, all such factors are incorporated into plume-dilution models, including simplified "screening-level" approaches.

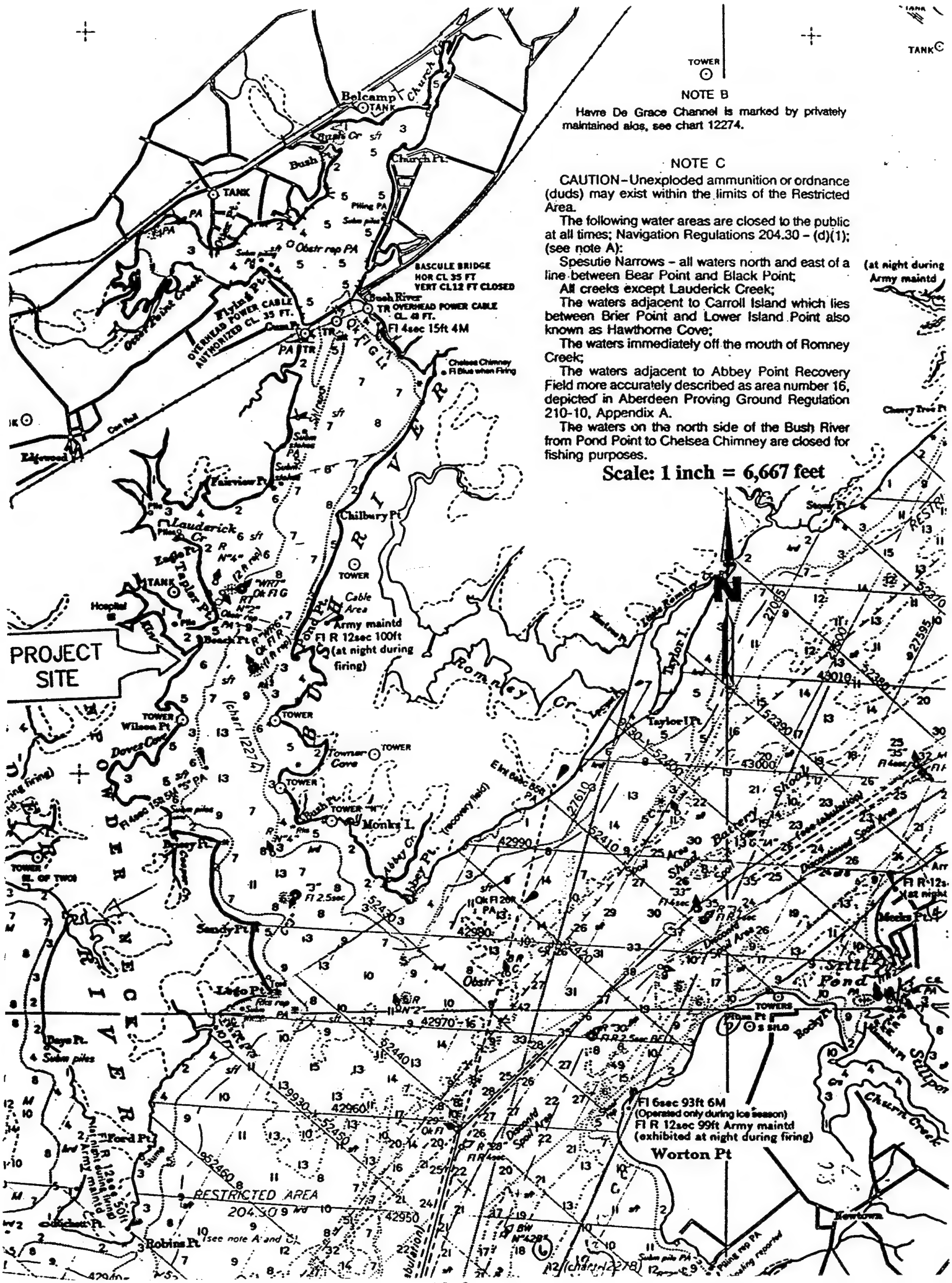
1.2 Objectives

The basic goal of this study is to estimate, using EPA-approved screening-level techniques, the initial ("near-field") dilution and flushing rate ("far-field" dilution) of groundwater discharge plumes within the adjacent surface waters. Such an assessment, when combined with relevant discharge concentration data, would provide first-order estimates of resulting near-field concentrations — receiving-water concentrations that would be expected over relatively short distances from the discharge site (e.g., ones, tens and hundreds of meters) and over short time periods (e.g., seconds and minutes). Our second objective is to compute such concentrations for selected toxic constituents and to compare the results to available baseline data. As a result, potential water quality impacts may be quantified. A third study goal is to assess the applicability of a "regulatory mixing zone" — a localized discharge zone in which local water quality standards may be exceeded. Such zones are routinely allowed for wastewater treatment plant discharges. If applicable to the subject groundwater-discharge plume, a mixing zone may allow for local exceedences of water quality standards.

1.3 Study Area

The goals and procedures outlined above are based on relevant characteristics of the study area. Surface elevations at the APG Beach Point site lie approximately 14.5 feet above sea level. An unsaturated zone occurs from elevation 14.5 ft down and at depths as shallow as 1 ft. The site is underlain by an impermeable clay lens located at a depth ranging from approximately 70 feet to 120 ft. A lower confined aquifer known as the Canal Creek Aquifer occurs below the upper confining unit.

The Bush River is a relatively small and shallow tributary estuary located on the western shore of the Upper Chesapeake Bay (Figure 1). The Bush River has a length of


NOTE B
Havre De Grace Channel is marked by privately maintained aids, see chart 12274.
NOTE C
CAUTION - Unexploded ammunition or ordnance (duds) may exist within the limits of the Restricted Area.
The following water areas are closed to the public at all times; Navigation Regulations 204.30 - (d)(1); (see note A):
Spesutie Narrows - all waters north and east of a line between Bear Point and Black Point;
All creeks except Lauderick Creek;
The waters adjacent to Carroll Island which lies between Brier Point and Lower Island Point also known as Hawthorne Cove;
The waters immediately off the mouth of Romney Creek;
The waters adjacent to Abbey Point Recovery Field more accurately described as area number 16, depicted in Aberdeen Proving Ground Regulation 210-10, Appendix A.
The waters on the north side of the Bush River from Pond Point to Chelsea Chimney are closed for fishing purposes.
Scale: 1 inch = 6,667 feet
PROJECT SITE
BASCULE BRIDGE
HOR CL 35 FT
VERT CL 12 FT CLOSED
OVERHEAD POWER CABLE AUTHORIZED CL. 35 FT.
Belcamp
Bush Pt.
Church Pt.
Lauderick Cr.
Fairview Pt.
Chilbury Pt.
Romney Cr.
Taylor I.
Wilson Pt.
Monks I.
Abbey Pt.
Sandy Pt.
Worton Pt
Robins Pt.
Ford Pt.
Still Pond
Churn Creek
Newtown
Edgewood
Hospital
Cable Area
Army maintd
Fl R 12sec 100ft (at night during firing)
Fl 6sec 93ft 6M
(Operated only during ice season)
Fl R 12sec 99ft Army maintd
(exhibited at night during firing)
RESTRICTED AREA
204.30
(see note A and C)
Discontinued Spoil Area
TOWER
TANK

approximately 14 km. Bush River MLW depths generally increase from an approximate 0.9-1.5 m range (near the head) to a 2.7-4.0 m range (near the mouth). The average Mean-Low-Water (MLW) depth is approximately 2 m. The average annual freshwater discharge to the Bush River from its 316-km^2 drainage area is only about 4.7 m^3/sec (Pritchard, 1976).

Bush River tides are predominantly semi-diurnal (NOAA, 1978), with a mean range at Pond Point of 0.4 m and a spring range of approximately 0.5 m (NOS, 1993a). Inside the entrance to the Bush River, about 2 miles south of Beach Point, average maximum tidal current speeds are approximately 0.3 m/sec (1.0 ft/sec; NOS, 1993).

The MLW volume of the Bush River is approximately 6.0×10^7 m^3; the inter-tidal volume (i.e., the "tidal prism") is about 1.2×10^7 m^3 (Pritchard, 1976). Thus, the inter-tidal volume comprises a relatively large fraction (approximately 20%) of the River's MLW volume. In contrast, the mean runoff volume per day (4.1×10^5 m^3/day) corresponds to a relatively small fraction of the River's MLW volume. Pritchard (1976) estimates that the MLW volume corresponds to approximately 150 days of mean freshwater inflow.

Due to its relatively small areal extent, depth, and freshwater inflow, the Bush River typically exhibits relatively small horizontal and vertical salinity gradients (Carter, 1976). For example, observations of system-wide salinity variations of only 3 ppt are reported in the Bush River (Martin Marietta, 1986; Carter, 1976). The longitudinal salinity difference is controlled primarily by the adjacent Chesapeake Bay salinities and secondarily by freshwater inflows to the Bush River from Otter Point Creek (Carter, 1976). Consequently, Bush River salinities do not always decrease monotonically in a

landward direction. Intratidal variations of Bush River salinities range from approximately 0.5 ppt near the mouth to near zero at the head (Carter, 1976).

Susquehanna River inflows to the Upper Chesapeake Bay influence the salinity distribution and circulation of the Bush River and other small tributary estuaries (Pritchard, 1976). As noted by Pritchard (1969), salinities of these embayments are controlled largely by the relatively low salinities of the Upper Bay and by Upper Bay variability. For example, Martin Marietta (1986) reports observations of Bush River salinities varying from 0.7 ppt (at the River's head-of tide) to 3.7 ppt (at River mouth), consistent with prevailing local Bay salinities. However, the salinity of the Upper Bay and its tributaries also varies in response to seasonal changes in Susquehanna River flows, as described by Pritchard (1976).

During the autumn, when Susquehanna River flows are relatively low, Upper Chesapeake Bay salinities are constant or slowly increasing. At this time, the local freshwater inflow to the Bush River may be sufficient to set up an estuarine type gravitational circulation, with a net outflow of fresher surface waters and net inflow of more saline bottom waters. During the spring freshet, when salinities of the Upper Bay decrease over time, corresponding salinities of the tributary embayments (e.g., the Bush River) are relatively high. Consequently, a reverse estuarine circulation pattern may develop, with a net bottom outflow of more saline water from the River to the Bay proper, and a net surface inflow from the Bay proper into the River. Such seasonal gravitational circulation patterns enhance the flushing of the Bush River and other tributary embayments. Thus, the available far-field dilution would be greater at these times.

The Bush River at the Beach Point site is classified as a *Designated Use II*: Shellfish Harvesting Waters (COMAR 26.08.02.02B) — an actual or potential area for shellfish propagation. Approximately one mile upstream from Beach Point, the Bush River classification changes to *Designated Use I*: Water Contact Recreation, and Protection of Aquatic Life (i.e., the basic swimmable/fishable classification).

Toxic water quality conditions in the Bush River are reported in a recent study of contaminants in Chesapeake Bay/Tributary sediments conducted by the Maryland Department of the Environment (Eskin et al., 1994). The MDE's toxics monitoring program includes a Bush River sampling station near Gum Point (Figure 1, 39°26'06"N, 76°14'36"W), approximately 5 km (3 miles) north of the Beach Point site. Annual (1988-91) sediment metals concentrations for this stations are reproduced below in Table 1; individual observations of sediment PAH and TOC for 1991 are reproduced in Table 2. In the tables, both the No Observable Effect Level (NOEL) and the Probable Effects Level (PEL) are listed for each parameter, where available. Such levels provide guidance regarding the potential toxicity of these parameters (MacDonald, 1993).

As indicated in the MDE report, most sediment concentrations observed in the Bush River fall above the NOEL but below the PEL. Thus, toxic effects due to each trace metal are unlikely (individually). However, compared to other tributary stations, the Bush River Nickel levels are somewhat high.

Where sediment quality guidelines are available, the data also suggest that toxicity due to individual PAH and TOC compounds is unlikely. In fact, most Bush River PAH concentrations observed in the MDE's 1991 survey fall below the corresponding NOELs.

TABLE 1: SEDIMENT METAL CONCENTRATIONS IN THE BUSH RIVER

Parameter	Guideline Limits (ppm)		Observed Concentrations (ppm)			
	NOEL	PEL	1988	1989	1990	1991
As	8	64	34.37	45.80	38.30	19.70
Cd	1	7.5	0.01	0.15	0.40	0.50
Cr	33	240	68.92	69.00	82.00	59.00
Cu	28	270	43.32	45.00	47.50	40.00
Pb	21	160	42.25	66.00	42.60	54.00
Hg	0.1	1.4	0.099	0.244	0.150	0.190
Ni	–	–	53.79	57.00	47.10	52.00
Zn	68	300	205.0	216.0	222.3	189.0

TABLE 2: SEDIMENT PAH AND TOC DATA FOR THE BUSH RIVER

Parameter	Guideline Limits (ppb)		Observed Concentrations (ppb)
	NOEL	PEL	
anthracene	85	740	75.0
fluoranthene	380	3,200	24.4
pyrene	–	–	224.9
benzo(A)anthracene	160	1,300	75.0
benzo(B)fluoranthene	–	–	187.4
benzo(K)fluoranthene	–	–	56.2
benzo(A)pyrene	230	2,500	112.5
benzo(G, H, 1)perylene	–	–	150.0

2. NEAR-FIELD MODEL STUDY

2.1 Approach

In this preliminary assessment, the impact of groundwater emanating from the APG site was evaluated based on the potential dilution available within the Bush River. Within the near-field, it is postulated that a groundwater plume intersects the channel of the Bush River and seeps into this River as a submerged buoyant plume. Based on the available geologic information about the APG site, it is also postulated that the discharge from the surface aquifer is limited to a relatively narrow band that parallels the interface between the Bush River and the subject APG site. As this groundwater plume is less dense (less saline) the adjacent estuarine water, it would rise and subsequently mix with the ambient receiving-waters. Thus, it is diluted through mechanisms of buoyancy-based entrainment and mixing associated with ambient currents.

These same physical mechanisms are known to dilute effluent plumes discharged from the outfall diffusers of wastewater treatment plants (WWTP) and electrical generation facilities. Thus, the dilution of groundwater emanating from the APG site can be conceptualized as the dilution of the discharge from a line diffuser. This approach allows the near-field impact of this groundwater discharge to be simulated using established plume-dilution models that have been developed primarily for such outfall diffusers.

The limitations of this approach are twofold. First, most plume-dilution models also simulate momentum-based entrainment, a phenomenon that has a negligible effect upon the groundwater plume. Therefore, the port configuration on the

conceptualized line diffuser must be set to minimize initial momentum. Secondly, the conceptualization of the groundwater discharge surface as a line diffuser will concentrate the plume's impact within the near-field condition. As this effect reduces the dimensions of the near-field plume and the calculated levels of dilution, the model result will provide a conservative estimate of field conditions.

2.2 Near-Field Model Selection

A series of EPA-approved, plume-dilution models for surface waters are documented in "Initial Mixing Characteristics of Municipal Ocean Discharges" by Muellenhoff et al. (EPA/600/3-85/073a, 1985) and by reviewed by Alam et al. (1982). These models provide a convenient and accepted choice of screening-level techniques. As noted above, such models are usually applied to discharges from line sources such as outfall diffuser manifold legs. In the present case, however, the subject discharges are emanating primarily from the adjacent stream bed.

While all EPA-approved plume models can be run, ULINE is preferred for dilution calculations since it is based directly on laboratory experiments and has proved it's usefulness in recent outfall studies conducted throughout the U.S.(e.g., Najarian Assoc., 1990). The ULINE model also neglects initial discharge momentum, consistent with the problem at hand.

Typically, outfall dilution studies are conducted under "critical" (minimum-dilution; Tetratech, 1982) conditions. These conditions are defined as lowest 10th-percentile ambient currents, and highest 10th-percentile ambient density stratification conditions. However, Maryland regulations require that the acute toxicity criteria be applied under conditions of mean low water and "minimum daily averaged

1-hour tidal velocity" (COMAR 26.08.02.05C); chronic criteria are applied under conditions of mean water level and average tidal velocity (COMAR 26.08.02.05D). Thus, the input data for the selected near-field model (ULINE) should reflect these required conditions.

To adapt ULINE to the study area, an input of site-specific data is required. Such model input data fall into three categories: (1) groundwater discharge parameters; (2) receiving-water parameters; and (3) outfall diffuser parameters. The selection of such parameters is described below:

2.3 Estimation of Groundwater Input Parameters

Information relating the site's local geology and hydrogeologic conditions was obtained from previous studies by Jacobs Engineering Group and K-V Associates, Inc. (1994). These studies indicate that subsurface flows within the surficial aquifer are bounded by a sub-surface clay layer and that the portion of the aquifer underlying the Beach Point peninsula is both homogeneous and isotropic. On-site data indicates that the dominate, long-term groundwater flow direction would be down-gradient towards the Bush River.

Based on K-V Associates measurements and Jacobs Engineering calculations (Jacobs Engineering Group, 1994), the average saturated thickness of the surficial aquifer at the Beach Point peninsula is approximately 18.8 m (61.8 ft). Also, the Bush River frontage along the subject peninsula is approximately 448 m (1,470 ft). Approximately 366 m (1,200 ft) of frontage is directed in a northeast direction; approximately 82 m (270 ft) of the northern peninsula terminus is directed in a

northwest direction. Thus, the cross-section area of the saturated aquifer that parallels the Bush River is approximately 8,423 m^2 (90,846 ft^2).

As conditions within the surficial aquifer are homogeneous and isotropic, horizontal velocity measurements at a single location may be used to evaluate the aquifer discharge. K-V Associates measured a 25-hour average horizontal velocity of 0.27 m/day (0.89 ft/day) at a 15-m deep well site located near the middle of the Beach Point Peninsula. This flow was directed 61.8° northeast of magnetic north, approximately 24° riverward from the main peninsula axis. Note that the above velocity measurement is a 24-hour average value (discounting short-term fluctuations due to tidal influences).

Jacobs Engineering estimates the groundwater seepage rate to be 0.0136 m^3/sec (41,509 ft^3/day). This estimate utilizes the directional component of the horizontal velocity that is *orthogonal* to the Bush River frontage. That is, the measured horizontal velocity (i.e., 0.27 m/day) is multiplied by a sin(24°) factor along the 366 m (1,200-ft) frontage:

Discharge = (0.27 m/day velocity)(sin24°)(18.8 m depth)(366 m frontage)
+ (0.27 m/day velocity)(18.8 m depth)(82 m frontage)

= 0.0136 m^3/sec

2.4 Ambient Receiving Water Input

Information relating the hydrographic conditions within the Bush River were obtained from a review of the NOAA Tidal Current Table data, NOAA navigation charts, and previous studies of the Bush River by Martin Marietta (1986).

Ambient Current Velocity in Bush River

A review of the NOAA Tidal Current Table data at a location of 0.6 km southwest of Bush Point indicates that the maximum flood velocity is 0.305 m/sec (1.0 ft/sec or 0.6 knot). Assuming a pure sinusoidal tidal velocity distribution, having a 0.305 m/sec amplitude and 12.4 hour period, the "minimum daily average 1-hour tidal velocity was computed. That is, the assumed tidal current magnitude profile was averaged over a 1-hour window centered at slack tide (approximately hour 6.2). The resulting speed was estimated to be 0.04 m/sec (see Appendix A) . Thus, as a conservative estimate, an ambient current speed of 0.03 m/sec was input to the near-field model as a representative acute condition tidal current speed. Likewise, for the same sinusoidal profile, a tidal average speed was computed as 0.19 m/sec. This latter average is representative of the chronic condition speed.

Ambient salinity and temperature in Bush River

A representative receiving-water salinity of 3 ppt is assumed in this study, consistent with the limited available data (e.g., Martin Marietta, 1986). Similarly, a temperature of 21° C is assumed for the Bush River. This salinity and temperature correspond to a receiving-water density of only 1.00031 g/cm^3. Thus the selected values are conservative, inasmuch as only limited plume buoyancy will be available to dilute the discharges. Note that the assumed density of the groundwater discharges is 1.00024 g/cm^3 (i.e., only 0.007% lighter than the ambient receiving waters).

Depth of discharge

Based on the most recent NOAA navigation chart for the Bush River, an average water depth of 1.5 m (5 ft) is assumed at the Bush River discharge site. However, a more conservative estimate of half this depth, 0.76 m (2.5 ft), is chosen as a representative model discharge depth. This selection reflects the fact that some of the discharge occurs along the shallow frontage of the peninsula.

2.5 Conceptualized Discharge Schematization

As noted above, the 448-m-wide groundwater discharge plume at the Beach Point site seeps into the adjacent Bush River. Thus, this discharge is assumed to comprise a 448-m-long line source of buoyancy. By analogy, the corresponding plume in the Bush River is assumed to be discharged from a 295-port diffuser having a total length of 448 m and a "port" spacing of 1.5 m (5 ft). As noted by Roberts et al. (1989), the dilution level of a buoyancy-dominated plume is nearly independent of the number of ports, and depends primarily on *plume length* and buoyancy. The schematized diameter of each port is 1.2 m (4 ft). Such large diameter ports allowed momentum-related effects to be minimized. In addition, further testing showed that Model results are insensitive to port diameter (see section 2.7).

2.6 Near-Field Model Results

Using information described above, an input file was constructed for the ULINE model and a series of simulation were conducted. *The results indicate that a near-field dilution ratio of* **159:1** *is appropriate for application of acute toxicity criteria. Similarly, model results indicate that a near-field dilution ratio of* **1,006:1** *is appropriate for application of chronic toxicity criteria.* Having limited buoyancy and initial

momentum, this near-field dilution is effected largely by the large discharge length of 427 m (1,400 ft) and ambient current field.

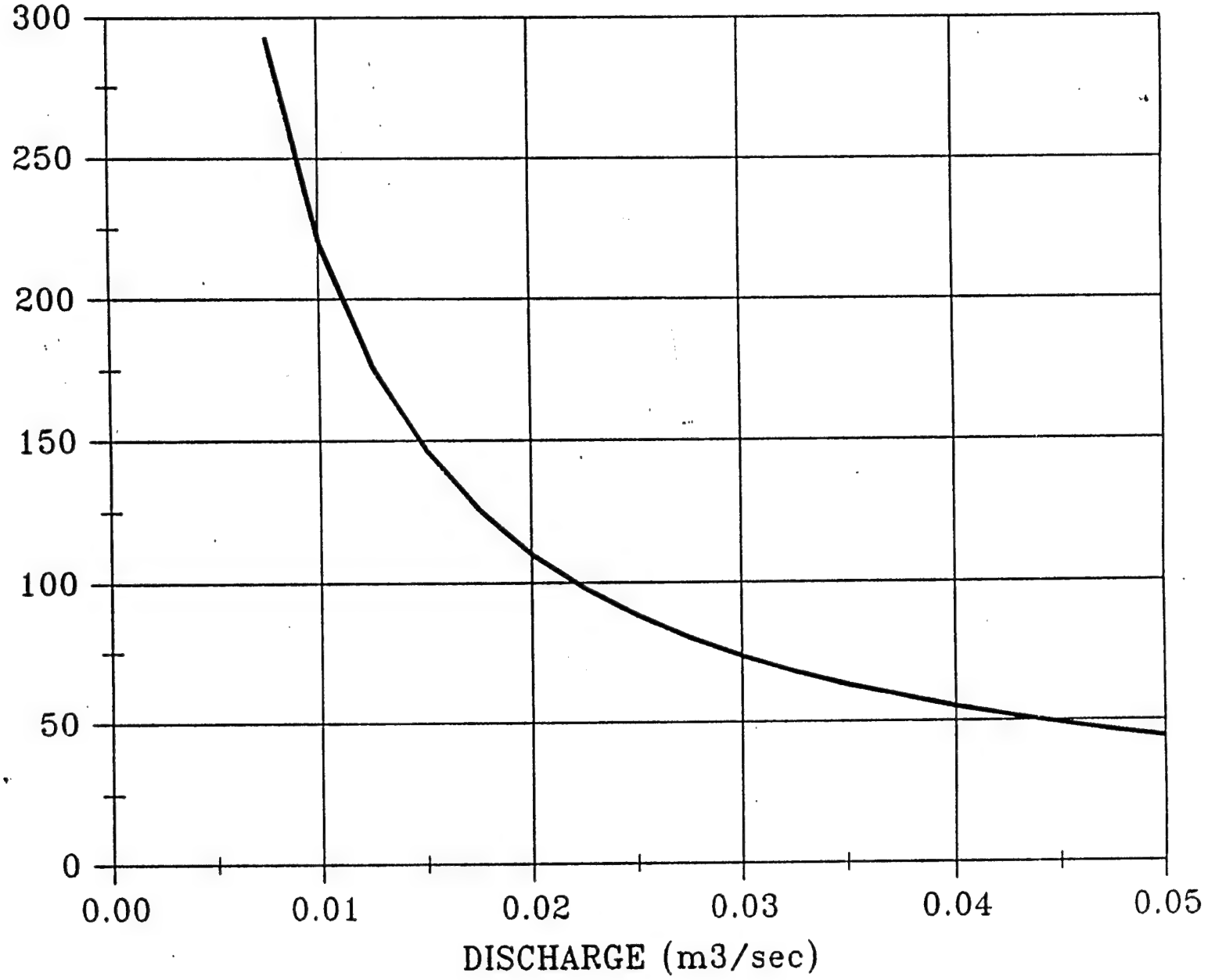
2.7 Model Sensitivity Analysis

The accuracy of any model results depends, in part, upon the accuracy of selected input parameter values. Accordingly, a model sensitivity analysis was performed to evaluate effects of individual parameter variations on model output. Results of this analysis are plotted in Figures 2-6 and described below.

Figure 2 displays model sensitivity to variations in the groundwater discharge to the Bush River. As noted above, the selected discharge rate of 0.0136 m^3/sec (0.48 cfs) corresponds to a dilution ratio of 159. However, higher discharge rates are likely to decrease the near-field dilution (Figure 2). For example, a near doubling of the discharge rate to 0.025 m^3/sec nearly halves the dilution to a ratio of only 88.

Likewise, Figure 3 represents the model sensitivity to the selected discharge length (i.e., the assumed length of the "line source"). As noted above, the discharge length is estimated to be 448 m (1,470 ft). This value assumes the extent of the lateral discharge to be the entire length of the Beach Point Peninsula fronting Bush River. However, a more conservative estimate of approximately half this length (i.e., 224 m, 735 ft) corresponds to a simulated near-field dilution ratio of 81. Also, a discharge of length of (152 m) 500 ft corresponds to a dilution ratio of 54. Overall, the dilution is somewhat sensitive to, and approximately linearly related to, this assumed discharge length.

SIMULATED DILUTION vs. DISCHARGE



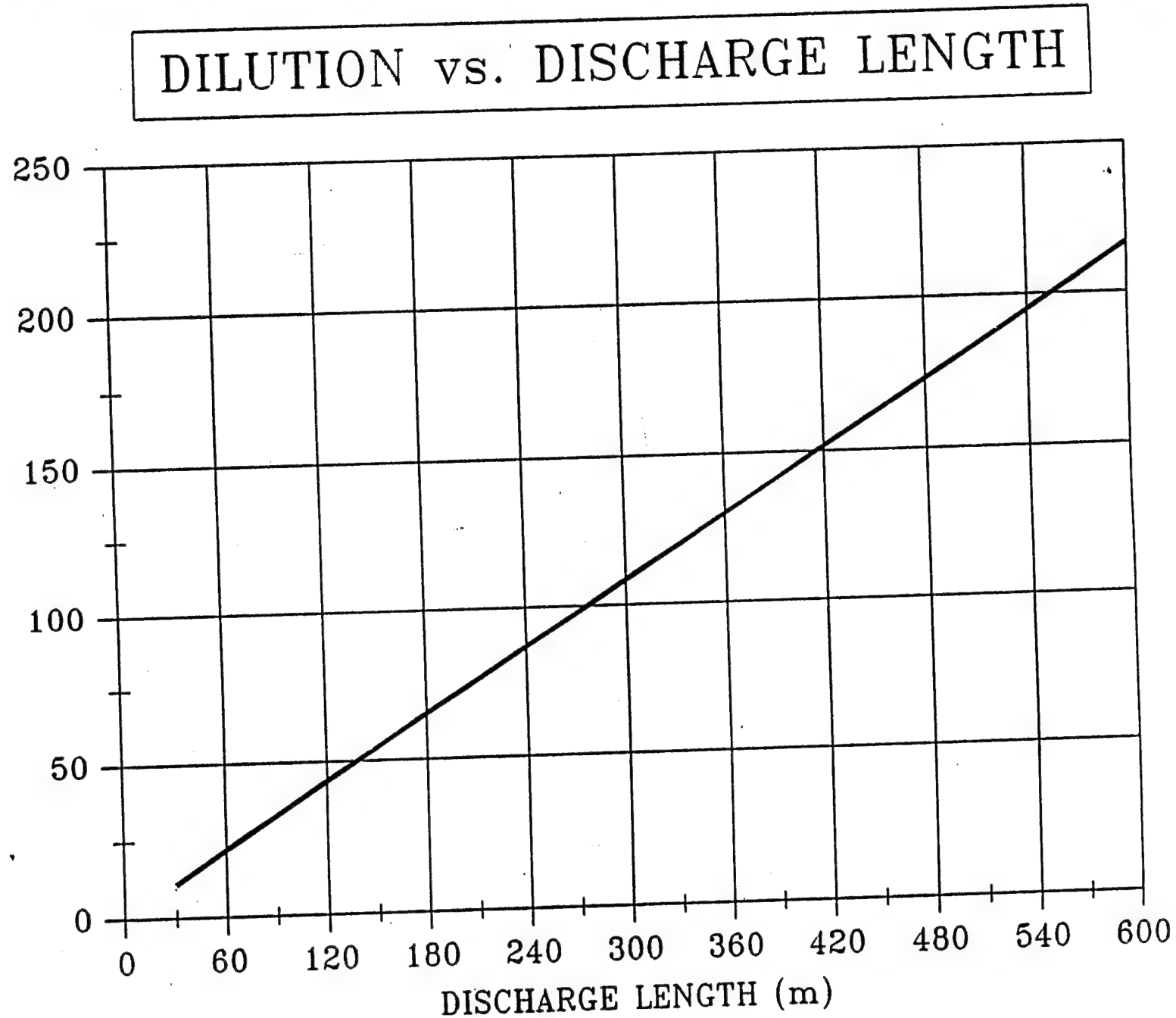

DILUTION vs. DISCHARGE LENGTH
250
200
150
100
50
0
DILUTION
0
60
120
180
240
300
360
420
480
540
600
DISCHARGE LENGTH (m)

FIGURE 3

The model results are also somewhat dependent upon the assumed discharge depth. Figure 4 shows a nearly linear relation between the simulated dilution ratio and discharge depth. In this figure, note that the average water depth in the near field is approximately 1.5 m (5 ft). However, a conservative value of half this depth (i.e., 0.75 m or 2.5 ft) is used in all other simulations. This conservative assumption results in a dilution of 159 — nearly half of the corresponding dilution for a 1.5-ft-deep discharge (i.e., 319).

Typically, near-field models are sensitive to the ambient current speed. Figure 5 suggests that the adapted model is no exception. For example, the estimated tenth-percentile speed (i.e., 0.03 m/sec) corresponds to a dilution ratio of 159, while a speed of 0.02 m/sec corresponds to a dilution of 106. Thus, it is unlikely that uncertainties in the speed estimate will greatly enhance the computed dilution levels.

Finally, the adapted model assumes a Bush River salinity of 3 ppt. Figure 6 suggests that the model is relatively insensitive to other representative values for this parameter.

Based on the overall results of the model sensitivity analysis, a more conservative estimate of the near-field dilution factor would lie within the range of 54 to 159 for application of acute toxicity criteria. Similarly, a conservative estimate of the near-field dilution factor would lie within the range of 499 to 1,006 for application of chronic toxicity criteria.

DILUTION vs. DISCHARGE DEPTH

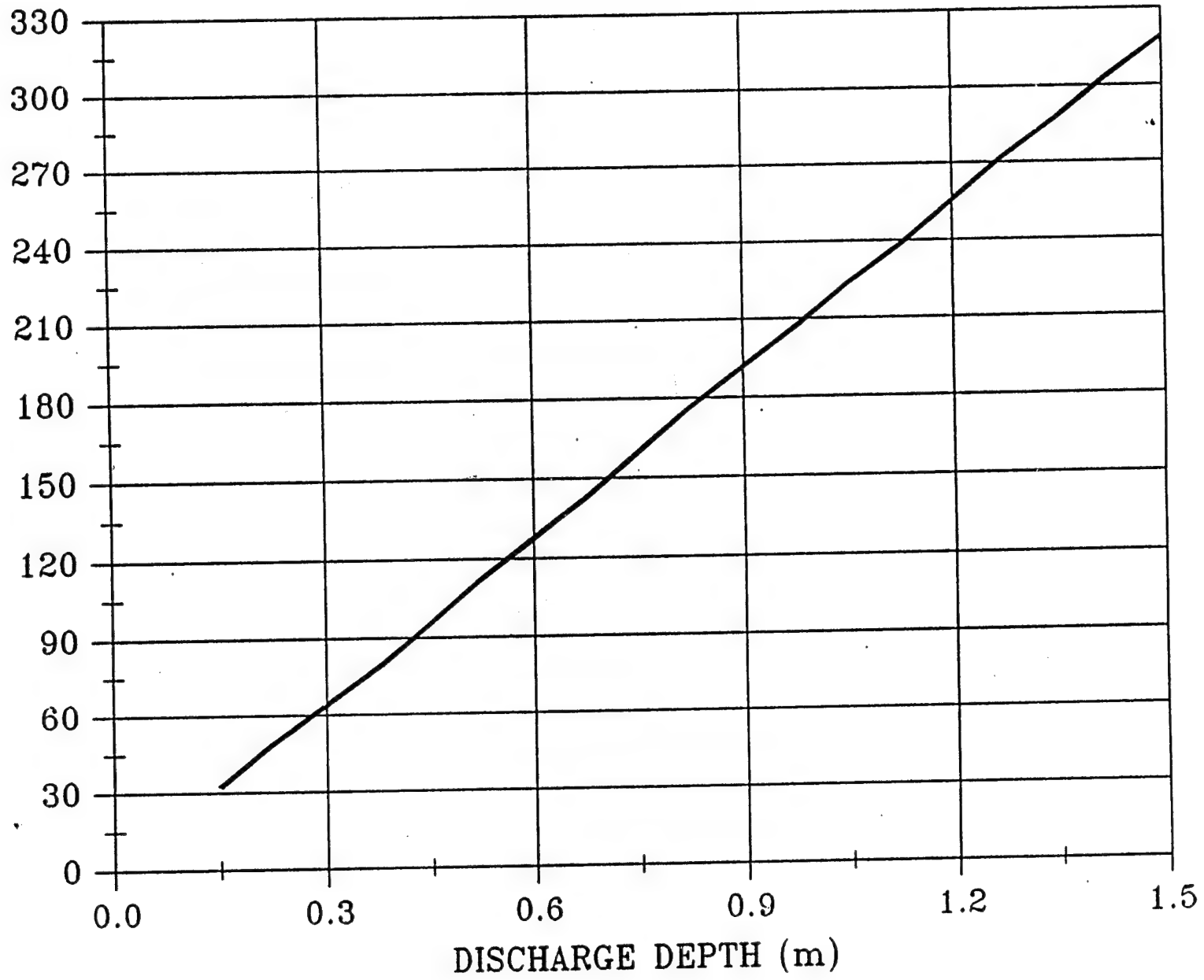


FIGURE 4

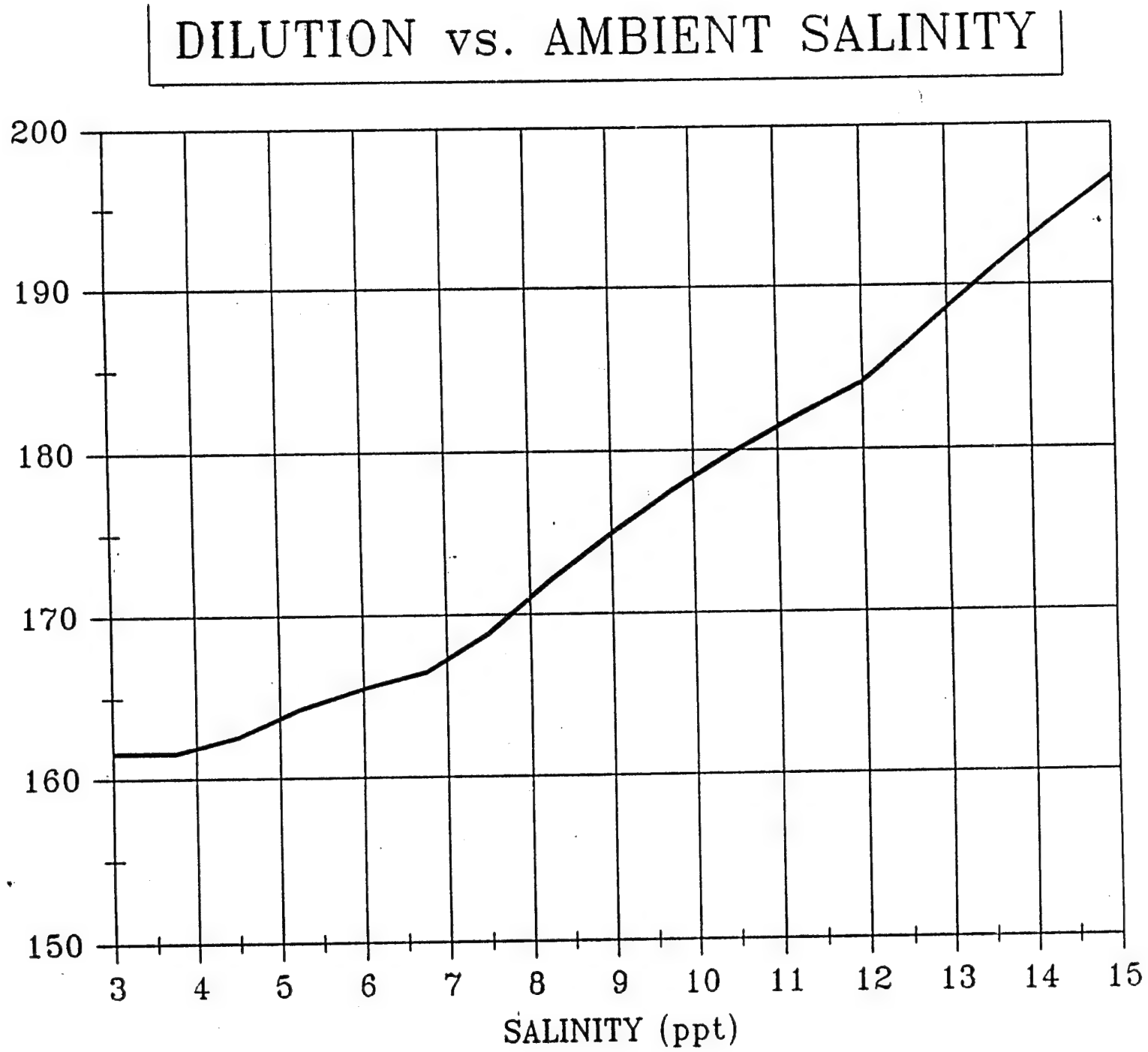

DILUTION vs. AMBIENT SALINITY
DILUTION FACTOR
200
190
180
170
160
150
3
4
5
6
7
8
9
10
11
12
13
14
15
SALINITY (ppt)

FIGURE 6

SIMULATED DILUTION vs. AMBIENT CURRENT

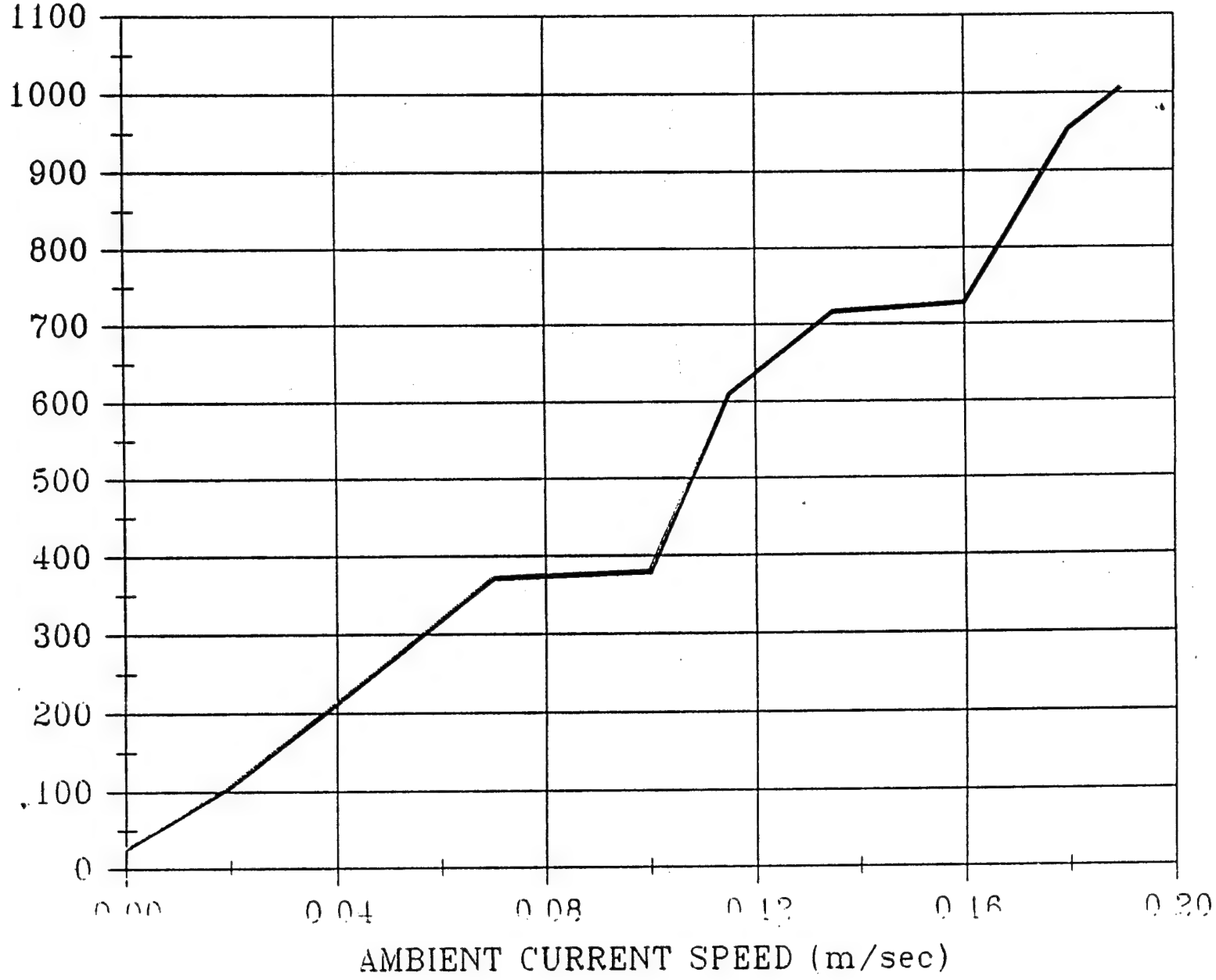


FIGURE 5

3. FAR-FIELD DILUTION ESTIMATES

3.1 Definitions

The Bush River receiving waters can be divided into two zones based on the physical mechanisms of mixing. In the near-field, rapid mixing is caused primarily by the discharge plume's buoyancy and initial discharge momentum, in combination with ambient currents. As noted above, near-field length scales are typically ones to tens of meters; near-field time scales are typically seconds to tens of minutes.

In the far-field, the plume is mixed passively by ambient processes such as turbulence and tidal dispersion. Far-field length scales are typically kilometer distances from the discharge site; far-field time scales are hours, days or more.

The following sections provide preliminary assessments of the far-field dilution (i.e., the flushing rate) in the Bush River Estuary. Methodologies for such techniques are documented in "Water Quality Assessment: a Screening Procedure for Toxic and Conventional Pollutants" (EPA/600/6-85/002b). Input required for these analyses includes the local tidal range, mean freshwater inflow and salinity data.

3.2 Tidal Prism Method

The tidal prism method was first introduced by Ketchum (1951) and later elaborated (EPA, 1985). This method divides an estuary into a number of hypothetical segments having lengths of approximately one tidal excursion. Within each segment, complete mixing is assumed. Thus, the flushing flow for the i^{th} segment, Q_{fi}, is given by the tidal prism volume for that segment, P_i, divided by the dominant tidal period, T (i.e., 1 tidal cycle = 12.42 hours or 44,712 sec). The

corresponding residence time for that segment is computed as the total segment volume (i.e., P_i+V_i, where V_i is the MLW volume) divided by the segment flushing rate. The total segment residence time is the sum of all individual segment residence times.

Application of this method to the Bush River Estuary is illustrated in Table 3. In this application, the Bush River is divided into 6 hypothetical segments as delineated in the previous one-dimensional modeling study by Carter (1976). Typical segments are approximately 2-3 km long, comparable to one tidal excursion. Table 3 lists corresponding intertidal volumes and MLW (i.e., "subtidal") volumes for each segment. Table 3 also lists the computed segment "exchange ratio," r_i, defined as the tidal prism, P_i, divided by the total segment volume, P_i+V_i. Thus, the computed residence time for the i^{th} segment (in units of tidal cycles) is simply the inverse of r_i. The tidal prism method calculations indicate that the total residence time for the Bush River is approximately 31 days. Moreover, the flushing time below the Beach Point site (i.e., for segments 2 and 1) is approximately 13 days (24.3 cycles). This suggests that the Bush River has limited "memory" of past discharges from the site. It should be noted, however, that the tidal prism method may underestimate the flushing time of an estuary since the selected segments are not, in reality, completely mixed.

Based on Table 3, the theoretical flushing rate at the Beach Point site, Q_{f2}, is given by the ratio P_2/T or approximately 1.0×10^6 m^3 per tidal cycle i.e., per 44,712 seconds). Thus, the corresponding far-field dilution is obtained by dividing this flushing rate by the assumed groundwater discharge flow, 0.0136 m^3/sec (see Eq. 6).

TABLE 3: CALCULATION TABLE FOR TIDAL PRISM METHOD

Segment Number	Segment Dimensions				Subtidal Water Volume, V_i (m^3)	Intertidal Water Volume P_i (m^3)	Segment Exchange Ratio r_i	Segment Flushing Time, T_i (Tidal Cycles)
	Starting Distance Above Mouth (m)	Ending Distance Above Mouth (m)	Distance of Center Above Mouth (m)	Segment Length (m)				
6	18,810	16,110	17,460	2,700	2.0×10^6	4.3×10^5	0.18	5.6
5	15,310	11,910	13,610	3,400	6.1×10^6	8.7×10^5	0.12	8.3
4	11,910	9,260	10,585	2,650	8.0×10^6	8.5×10^5	0.10	10.0
3	9,260	6,560	7,910	2,700	1.1×10^7	9.5×10^5	0.08	12.5
2	6,560	3,700	5,130	2,860	1.3×10^7	1.0×10^6	0.07	14.3
1	3,700	0	1,850	3,700	1.6×10^7	1.7×10^6	0.10	10.0
							$\sum_{i=1}^{n} T_i =$	60.7 cycles (31.4 days)

Down Estuary

This yields an estimated far-field dilution of 1,618. Again, note that this method likely overestimates this dilution since it assumes complete mixing within segments.

3.3 Fraction of Freshwater Method

This alternate method for estimating estuarine flushing times uses the freshwater content of estuarine segments as a tracer. That is, the "fraction of freshwater" for the i^{th} segment, f_i, is the defined as the difference between the Bush River entrance salinity and the i^{th} segment salinity, S_s-S_i, divided by the entrance salinity, S_s (see Table 4, sixth column). The corresponding "freshwater volume," W_i, for that segment is approximated by the fraction of freshwater, f_i, multiplied by the mean segment volume, V_i. Thus, in the steady state, the segment flushing time is simply the segment freshwater volume divided by the freshwater inflow, R. Again, the total residence time is obtained by summing the individual segment flushing times.

Application of this method to the Bush River Estuary is illustrated in Table 4. Note that the segmentation used for this method is the same as that employed in the previous section. The fraction-of-freshwater-method calculations estimate the total residence time for the Bush River as 57 days — somewhat longer than the tidal prism method estimate. However, the computed flushing time below the Beach Point site (i.e., for segments 2 and 1) is nearly the same — approximately 14 days (26.7 cycles). This also suggests that the Bush River has limited "memory" of past discharges from the site.

3.4 Previous Dye-Tracer Study Results

Far-field dilution also may be computed empirically based on field experiments. Accordingly, the dye-tracer data collected by Carter (1976) in the Bush Riyer allows

TABLE 4: CALCULATION TABLE FOR FRACTION OF FRESHWATER METHOD

Segment Number	Mean Segment Salinity S_i (ppt)	Mean Segment Length (m)	Mean Segment Cross-sectional Area (m^2)	Segment Mean Tide Volume V_i (m^3)	Fraction of River Water $f_i = \frac{Ss-Si}{Ss}$	River Water Volume $W_i = f_i \times V_i$ (m^3)	Segment Flushing Time $T_i = W_i/R$ (tidal cycles)
1	4.3	3,700	4,865	1.8×10^7	0.10	1.8×10^6	8.6
2	3.5	2,860	4,895	1.4×10^7	0.27	3.8×10^6	18.1
3	2.9	2,700	4,444	1.2×10^7	0.40	4.8×10^6	22.8
4	1.9	2,650	3,320	8.8×10^6	0.60	5.3×10^6	25.2
5	1.1	3,400	2,053	7.0×10^6	0.77	5.4×10^6	25.7
6	0.7	2,700	889	2.4×10^6	0.85	2.0×10^6	9.5
						$\sum_{i=1}^{n} T_i =$	109.9 cycles (56.9 days)

Up Estuary

for such computations. Under spring (high-flow) conditions, Carter reports a steady dye dilution ratio of 267 (i.e., 61.4 ppb effluent concentration to 0.23 ppb measured). Likewise, under fall (low-flow) conditions, Carter reports a dilution ratio of 315 (i.e., 50.5 ppb initial concentration to 0.16 ppb measured). Carter's data provides a more conservative estimate of the far-field dilution than the segmented (complete mixing) approaches described in sections 3.2 and 3.3. For this reason, and because it is based on direct measurements, the above-mentioned dilution ratios of 267 and 315 are adopted for use in the calculation of total near-field dilution.

3.5 Calculation of Total Near-Field Dilution

For a conservative constituent, the far-field, near-field and total near-field dilutions (S_f, S_n and S_t, respectively) are defined (e.g., Fischer, 1979; Adams, 1990) as:

$$S_f = \frac{C_o - C_a}{C_f - C_a} \qquad (1)$$

$$S_n = \frac{C_o - C_f}{C_n - C_f} \qquad (2)$$

$$S_t = \frac{C_o - C_a}{C_n - C_a} \qquad (3)$$

where C_o, C_n and and C_f are the constituent concentrations at the discharge, at the end of the near-field, and in the far-field, respectively. Note that in these definitions, the far-field concentration serves as a background concentration for the near-field in the same way that the ambient concentration serves as a background for the total near-field. These definitions may be combined to yield (Adams, 1990):

$$S_t = \frac{1}{\frac{1}{S_n} + \frac{1}{S_f} - \frac{1}{S_n S_f}} \qquad (4)$$

or

$$\frac{1}{S_t} \sim \frac{1}{S_n} + \frac{1}{S_f} \qquad (5)$$

This inverse sum indicates that the total dilution in the near field is always less than both the near-field and far-field dilutions. Moreover, the effectiveness of near-field mixing is always limited by the magnitude of the far-field mixing.

In the absence of other pollutant sources (i.e., $C_a=0$ in Eq. 1), the far-field dilution reduces to a ratio of either concentrations or flows (Adams, 1990):

$$S_f = \frac{C_o}{C_f} = \frac{Q_f}{Q_o} \qquad (6)$$

where: C_o is the discharge concentration; C_f is the far-field concentration; Q_o is the discharge flow; and Q_f is the far-field "flushing flow" or downestuary transport rate (Officer, 1976). Note that the flushing flow is not a measurable flow rate, but is the equivalent rate at which mass is transported seaward.

Based on an application of Eq. 5 with a value of $S_n=50$, the total near-field dilutions for acute toxicity conditions is calculated to be approximately 42 and 43 for under spring and fall conditions, respectively. Similarly, an application of Eq. 5 with a value of $S_n=450$, yield total near-field dilutions for chronic toxicity conditions of approximately 168 and 185 for under spring and fall conditions, respectively.

3.6 Application of Dilution Factors to Toxics Discharge Data

Using the above calculated dilution factors, near-field concentrations can be estimated for the parameters of interest. This estimate would be equal to the mean groundwater

concentration divided by the dilution factor. Thus, using the above dilution factor, a pollutant concentration of 42 ppb in the groundwater would be diluted to 1 ppb within the near-field limits of the Bush River. In this manner, near-field concentrations for all detected groundwater constituents are calculated. These results concentrations are shown in Tables 5 and 6. An examination of these Table reveals that the expected near-field impacts will be near or below the method detection limit for most parameters. For such constituents, these analyses suggest that on-site contamination will not produce any discernable effect on the water quality of the Bush River.

Table 5: Computed Dilutions of Volatile Organics (ug/l)

Constituent	Groundwater Concentration	Receiving-Water Concentration	
		Acute*	Chronic**
Chlorobenzene	2	0.05	0.01
Chloroform	12	0.29	0.07
1,1-Dichloroethene	5	0.12	0.03
cis-1,2-Dichloroethene	640	15.24	3.81
1,1,1,2-Tetrachloroethane	3	0.07	0.02
1,1,2,2-Tetrachloroethane	17,000	404.76	101.19
Tetrachloroethene	90	2.14	0.54
1,1,2-Trichloroethane	110	2.62	0.65
Trichloroethene	1,600	38.10	9.52
Vinyl Chloride	2	0.05	0.01

* Based on observed average concentration and dilution ratio of 42:1
** Based on observed average concentration and dilution ratio of 168:1

Table 6: Computed Dilutions of Heavy Metals (ug/l)

Constituent	Groundwater Concentration	Receiving-Water Concentration	
		Acute*	Chronic**
Aluminum	1,500	35.71	8.93
Antimony	5	0.12	0.03
Arsenic	7	0.17	0.04
Beryllium	5	0.12	0.03
Boron	70	1.67	0.42
Cadmium	1.4	0.03	0.01
Chromium	5	0.12	0.03
Cobalt	110	2.62	0.65
Copper	8	0.19	0.05
Lead	5	0.12	0.03
Mercury	0.5	0.01	0.003
Molybdenum	40	0.95	0.24
Nickel	180	4.29	1.07
Selenium	5	0.12	0.03
Silver	1	0.02	0.006
Thallium	5	0.12	0.03
Tin	100	2.38	0.60
Zinc	310	7.38	1.85

* Based on observed average concentration and dilution ratio of 42:1

** Based on observed average concentration and dilution ratio of 168:1

4. MIXING ZONE CONSIDERATIONS

The concept of mixing zones is normally applied in the case of wastewater facilities and power plants that discharge to tidal/coastal waterways via an outfall diffuser manifold. The following sections summarize Maryland's mixing zone regulations and explore their applicability to the subject groundwater discharges.

4.1 Mixing Zone Regulations

A mixing zone is defined (COMAR 26.08.01.01B) as "an area contiguous to a discharge where surface water quality or groundwater quality does not have to meet: (a) all water quality criteria; or (b) all requirements otherwise applicable to the natural water." That is, pollutant concentrations within the mixing zone may exceed the specified water quality standards within a localized vicinity of the outfall. The dimensions of this zone are normally comparable to those associated with the initial dilution processes (e.g., 10-100 meters). Thus, marine organisms would only be exposed to concentrations exceeding the specified criteria for a brief period during a transit of the mixing zone (Muellenhoff et al., 1985). Of course, water quality criteria must still be met outside this zone. Moreover, as stated in COMAR 26.08.02.05A, mixing zones may be allowed "provided that the following requirements are met outside the mixing zone: (a) there shall be no interference with biological communities or populations of indigenous species to a degree which is damaging to the aquatic life or ecosystem; (b) there shall be no diminishing of other legitimate beneficial uses; (c) mixing zones may not form barriers to the migratory routes of aquatic life; (d) mixing zones shall be designated and located to protect surface waters and shallow water shoreline areas; (e) the general water

quality criteria set out in regulation .03B(1)-(3) of this chapter apply within this mixing zone."

Four types of numerical toxic substance criteria are promulgated by the MDE: (a) human health criteria through ingestion of public water supplies; (b) the wholesomeness of fish for human consumption; (c) fresh, estuarine, and salt water aquatic life criteria from *acute* toxicity impacts; and (d) fresh, estuarine, and salt water aquatic life criteria from *chronic* toxicity impacts. For the purpose of applying numerical toxic substance criteria, the Bush River Area (Sub-Basin 02-13-07) is classified by the MDE as "fresh water" (Comar 26.08.02.03-1B) and as "Use II" (shellfish harvesting waters). Corresponding toxic substances criteria for both ambient surface waters ("fresh water") and human health ("fish consumption") are listed in Table 7. Note that toxic criteria related to drinking water do not apply within shellfish harvesting waters.

The following regulations (COMAR 26.08.02.05D) pertain to the application of toxic substance *Chronic Criteria* for the protection of Aquatic Life. First, in terms of size, the Regulatory Mixing Zone (i.e., the mixing zone in which the Chronic Criteria are applied) may not exceed 10 percent of the cross-sectional area of the receiving waters. Also, *the Chronic Criteria are applied under flow conditions determined from site-specific data for the mean tide level, average tidal velocity, and, when appropriate, the design streamflow.* Based on the MDE's discretion, a plume-dilution study also may be required for the application of these criteria. In accordance with COMAR 26.08.02.06(a), conditions pertinent to the application of toxic substance *Chronic Criteria* are also relevent to the application of toxic substance *Human Health Criteria.*

Table 7: Applicable Toxic Substances Criteria for the Bush River (ppb)

Constituent	Aquatic Life		Human Health
	Acute	Chronic	Fish Consumption
Arsenic (tri)	360	190	
Cadmium	3.9	1.1	
Chromium (hex)	16	11	
Chromium (tri)	1700	210	3,433,000
Copper	18	12	
Cyanide	22	5.2	
Lead	82	3.2	
Mercury	2.4	0.012	0.146
Nickel	1,400	160	100
Selenium	20	5	
Silver	4.1	0.12	
Zinc	120	110	
Aldrin	3		0.00079
Benzidine			0.0053
1,1DCE			18.5
DDT	1.1	0.001	0.00024
Dieldrin	2.5	0.0019	0.00076
Endrin	0.18	0.0023	
Lindane	2	0.08	0.625
PCP	20	13	
PCB	2	0.014	0.00079
1,1,1TCA			1,030,000
Toxaphene	0.73	0.0002	0.0073
TCE			807
Tributyltin (TBT)		0.026	
2,3,7,8 TCDD (Dioxin)			0.0000012

The following regulations (COMAR 26.08.02.05C) pertain to the application of toxic substance *Acute Criteria* for the protection of Aquatic Life. Such acute criteria (for low-velocity dischargers) must be met within a short distance from the outfall using the most restrictive of the following conditions:

(a) Within 10 percent of the distance (in any spatial direction) from the discharge to the edge of the Regulatory Mixing Zone (RMZ) used for the application of the chronic criteria.

(b) Within a distance of 50 times the "discharge length scale" (i.e., 50 times the square root of the cross-sectional area of the discharge outlet).

(c) Within a distance of 5 times the local water depth in any horizontal direction from the discharge outlet.

Thus, the region where the acute criteria are applied, sometimes called the "Toxic Dilution Zone" (TDZ), extends outward from the discharge to a distance no larger than any of the three lengths cited above. Moreover, the TDZ must not occupy more than 5 percent of the cross-sectional area of the receiving waters.

Also, *the acute criteria are applied under flow conditions determined from site-specific data for the mean low water elevation, the minimum daily averaged 1-hour tidal velocity and, when appropriate, design streamflow conditions*. Such tidal stages are deemed "critical" (minimum-dilution) conditions.

4.2 Site Applicability

The regulatory conditions relevent to mixing zones were developed for outfall diffusers. Their application to an area-wide groundwater discharge is somewhat uncertain due to the unknown extent of this area. It seems likely that the discharge area is confined within the the RMZ, which for this case extends approximately 500 feet from the river

bank. However, the extent of the TDZ is more poorly defined as it partially depends on the configuration of the discharge. Thus, the applicability of the mixing zone concept must be further examined.

The concept of mixing zones provides an intermediate approach to the assessment of toxic constituent impacts. The two other approaches generally in use are "end of pipe" and "fully mixed". The "end of pipe" approach allows for zero dilution -- criteria are directly applied to the discharge water quality. This simplified approach yields the most stringent results which, in turn, most ensures the protection of aquatic biota. However, it may lack a physical or biological basis in terms of receiving water impacts. Thus, its results are not considered to be defensible -- it is effectively a non-technically based policy decision. Conversely, the "fully mixed" (or far-field) approach considers the total volume of the receiving water segment to be instantly available for the dilution of a discharge. This approach is generally is far less stringent than the "end of pipe" approach. However, it may not adequately address biological impacts in the receiving water due to near-field conditions. The concept of mixing zones was evolved to address these limitations. Mixing zones allow consideration of the initial dilution process. This approach also protects aquatic life by limiting the potential for exposure to elevated concentrations to a minor portion of the waterway. As only a portion of the available dilution is considered in this approach, mixing zones provide a much more stringent basis for regulation than the "fully mixed "approach.

In the application of the mixing zone concept to the Bush River, it was necessary to conceptualize the area-wide groundwater discharge from the APG as a line source (diffuser). Due to this approach, potential impacts could be assessed in a technically defensible manner. However, this approach had a side effect of concentrating the

groundwater impact into a smaller area. *That is, this approach minimized the available near-field dilution and projected higher receiving water concentrations than would actually occur. Thus, the mixing zone analysis produced an overly conservative result. In the near-field, actual receiving water concentrations would be less than those projected due to more dispersed nature of the groundwater plume.* In the far-field, projected and actual receiving water concentrations should be comparable. However far-field concentrations would always be less than the near-field projections.

4.3 Conformance with Criteria

In Tables 8 and 9, the anticipated near-field receiving water concentrations are compared with the applicable water quality criteria. As shown in these Tables, no exceedences of these criteria are projected.

There will be a detectable impact on the receiving water concentrations due to volatile organic constituents (such as cis-1,2 dichloroethene; 1,1,2,2 tetrachloroethane; and trichloroethene). However, with the exception of trichloroethene (TCE), Maryland has not implemented criteria for these constituents.

Table 8: Comparison of Projected Near-Field Concentration with Applicable Toxic Substances Acute Criteria (ppb)

Constituent	Near-Field Concentration	Aquatic Life Acute Criteria
Arsenic (tri)	0.17 (total)	360
Cadmium	0.03	3.9
Chromium (hex)	0.12 (total)	16
Chromium (tri)		1700
Copper	0.19	18
Cyanide		22
Lead	0.12	82
Mercury	0.01	2.4
Nickel	4.29	1,400
Selenium	0.12	20
Silver	0.02	4.1
Zinc	7.38	120
DDT		1.1
Dieldrin		2.5
Endrin		0.18
Lindane		2
PCP		20
PCB		2
Toxaphene		0.73

Table 8: Comparison of Projected Near-Field Concentration with Applicable Toxic Substances Chronic and Human-Health Criteria (ppb)

Constituent	Near-Field Concentration	Aquatic Life Chronic Criteria	Human-Health Fish Consump.
Arsenic (tri)	0.04 (total)	190	
Cadmium	0.01	1.1	
Chromium (hex)	0.03 (total)	11	
Chromium (tri)		210	3,433,000
Copper	0.05	12	
Cyanide		5.2	
Lead	0.03	3.2	
Mercury	0.003	0.012	0.146
Nickel	1.07	160	100
Selenium	0.03	5	
Silver	0.006	0.12	
Zinc	1.85	110	
Aldrin			0.00079
Benzidine			0.0053
1,1DCE			18.5
DDT		0.001	0.00024
Dieldrin		0.0019	0.00076
Endrin		0.0023	
Lindane		0.08	0.625
PCP		13	
PCB		0.014	0.00079
1,1,1TCA			1,030,000
Toxaphene		0.0002	0.0073
TCE	9.52		807
Tributyltin (TBT)		0.026	
2,3,7,8 TCDD (Dioxin)			0.0000012

5. SUMMARY AND CONCLUSIONS

This study was initiated to assess the potential impact to the Bush River due to contaminated groundwater emanating from the APG site in Harford County, Maryland. Towards this objective, an EPA-approved near-field model was adapted to the receiving-waters adjacent to the APG site. The model approach and input was developed to provide a conservative assessment of receiving water impacts. This analysis suggests a near-field dilution level of approximately 42:1 for the application of acute criteria and a near-field dilution level of approximately 168:1 for the application of chronic criteria. Thus, contaminants introduced into Bush River receiving-waters at a concentration of 42 ppb will be diluted locally to a concentration of approximately 1 ppb or less. By applying these dilution factors to the groundwater quality data from the APG site, near-field concentrations within the Bush River can be projected. As shown in Tables 8 and 9, none of the projected receiving water concentrations exceed Maryland's current acute or chronic criteria.

There will be a detectable impact on the receiving water concentrations of certain volatile organic constituents (such as cis-1,2 dichloroethene; 1,1,2,2 tetrachloroethane; and trichloroethene). Should Maryland enact more stringent criteria for these parameters, these results and conclusions must be re-evaluated. In such a case, a field study would be recommended. Such a study would include in-situ current measurement, dye release and tracking, CTD casts, and/or water quality data collection.

REFERENCES

Adams, E.E., 1990. "Analysis of initial mixing in the Hackensack River near the outfall for the Bergen County Utility Authority wastewater Treatment Facility." Report prepared for Clinton Bogert Associates, Englewood Cliffs, New Jersey, 88 pp.

Adams, E.E., Stalzenbach, K.D., and Harleman, D.R.F., 1975, "Near and Far Field Analysis of Thermal Discharges into Large Bodies Of Water," Technical Report No. 205, R.M. Parsons Laboratory for Water Resources and Hydrodynamics, MIT.

Alam, A.M., Harleman, D.R.F.,and Colonell, J.M., 1982. "Evaluation of selected initial dilution models," *ASCE, Jour. Envir. Engnr. Div.,* 108:159-186.

Carter, H.H., 1976. "Simple one dimensional kinematic model results for the Bush River and Romney Creek." Chesapeake Bay Institute, Special Report 49, Reference 76-2, 37 pp.

Eskin, R.A., Rowland, K.H., Alegre, D.Y., and Magnien, R.E., 1994. "Contaminants in Chesapeake Bay sediments (1984-1991)." Chapter 4 of Draft Report, Maryland Department of the Environment, June 1994, 164 pp.

Fischer, H.B., List, E.J., Koh, R.C.Y., Imberger, N.H., and Brooks, N.H., 1979, Mixing in Inland and Coastal Waters, Academic Press.

K-V Associates, Inc., 1994. "Hydrogeologic study of Beach Point, Edgewood Area Aberdeen Proving Grounds, Maryland." Report prepared for Jacobs Engineering Group, Bel Air Maryland, 15 pp.

K-V Associates, Inc., 1994. "Estimated discharge rates in the surficial aquifer, Beach Point (Peninsula) study area, Aberdeen Proving Ground - Edgewood Area, Maryland." Letter report calculations, 1 p.

MacDonald, 1993. "Development of an approach to the assessment of sediment quality in Florida Coastal Waters." Report, Florida Department of Environmental Regulations.

Martin Marietta Environmental Systems, 1986. "Prediction of the effects of nutrient loadings from a power plant at Perryman, MD on the water quality

of the Bush River Estuary." Report prepared for the Maryland Department of Natural Resources, Power Plant Research Program, Annapolis Maryland.

Muellenhoff, W.P., et al, 1985, "Initial Mixing Characteristics of Municipal Ocean Discharges," Pacific Division, Environmental Research Laboratory, U.S. EPA, Newport, Oregon.

Najarian & Associates, Inc., 1990. "A Field and Model Study of the Dilution Capacity of the Passaic Valley Sewerage Commissioner's Wastewater Outfall System." Technical Report prepared for Passaic Valley Sewerage Commissioners, Newark, New Jersey.

Officer, C.B., 1976. Physical Oceanography of Estuaries and Associated Coastal Waters, John Wiley and Sons.

Pritchard, D.W., 1976. "Distribution of contaminants and excess heat in the Bush River and Romney Creek from the proposed Perryman Electric Power Plant." Chesapeake Bay Institute, Special Report 58, Reference 76-13, 30 pp.

Roberts, P.J.W., Snyder, W.H., and Baumgartner, D.J., 1989. "Ocean outfalls. I: Submerged wastefield formation." *ASCE, Jour. Envir. Engnr. Div.,* 115:1-25.

Roberts, P.J.W., Snyder, W.H., and Baumgartner, D.J., 1989. "Ocean outfalls. II: Spatial Evolution of Submerged Wastefield." *ASCE, Jour. Envir. Engnr. Div.,* 115:26-48.

Roberts, P.J.W., Snyder, W.H., and Baumgartner, D.J., 1989. "Ocean outfalls. III: Effect of diffuser design on submerged wastefield." *ASCE, Jour. Envir. Engnr. Div.,* 115:49-71.

Tetra Tech, Inc, 1982, "Design of 301(h) Monitoring programs for Municipal Wastewater Discharges to Marine Waters," EPA 430/9-82-010.

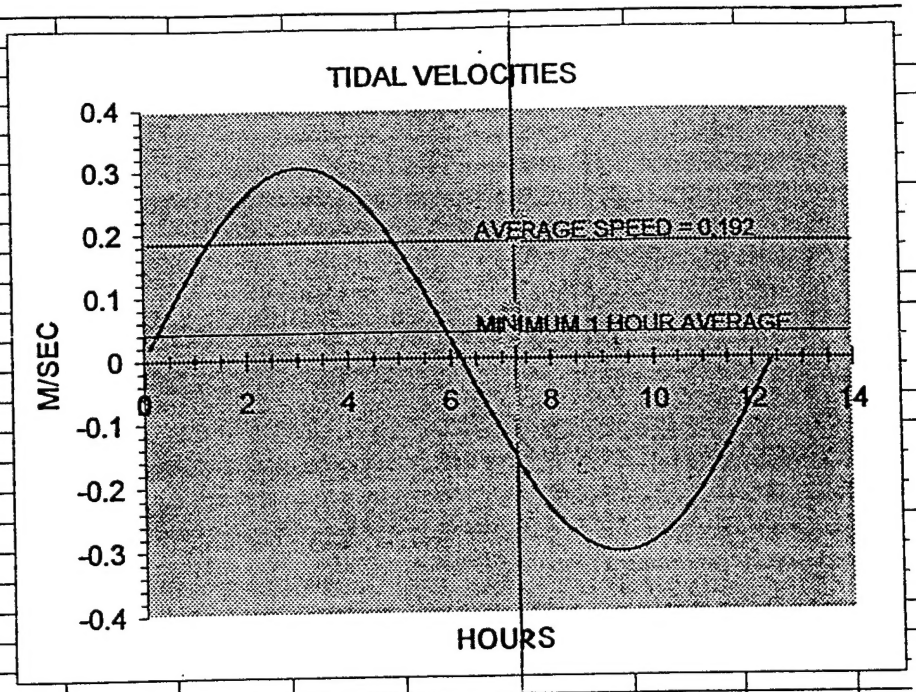
Tetra Tech, Inc, 1982, "Revised Section 301(h) Technical Support Document", EPA 430/9-82-011.

USEPA, 1985, "Technical Support Document for Water Quality-based Toxics Control," EPA-440/4-85-032.

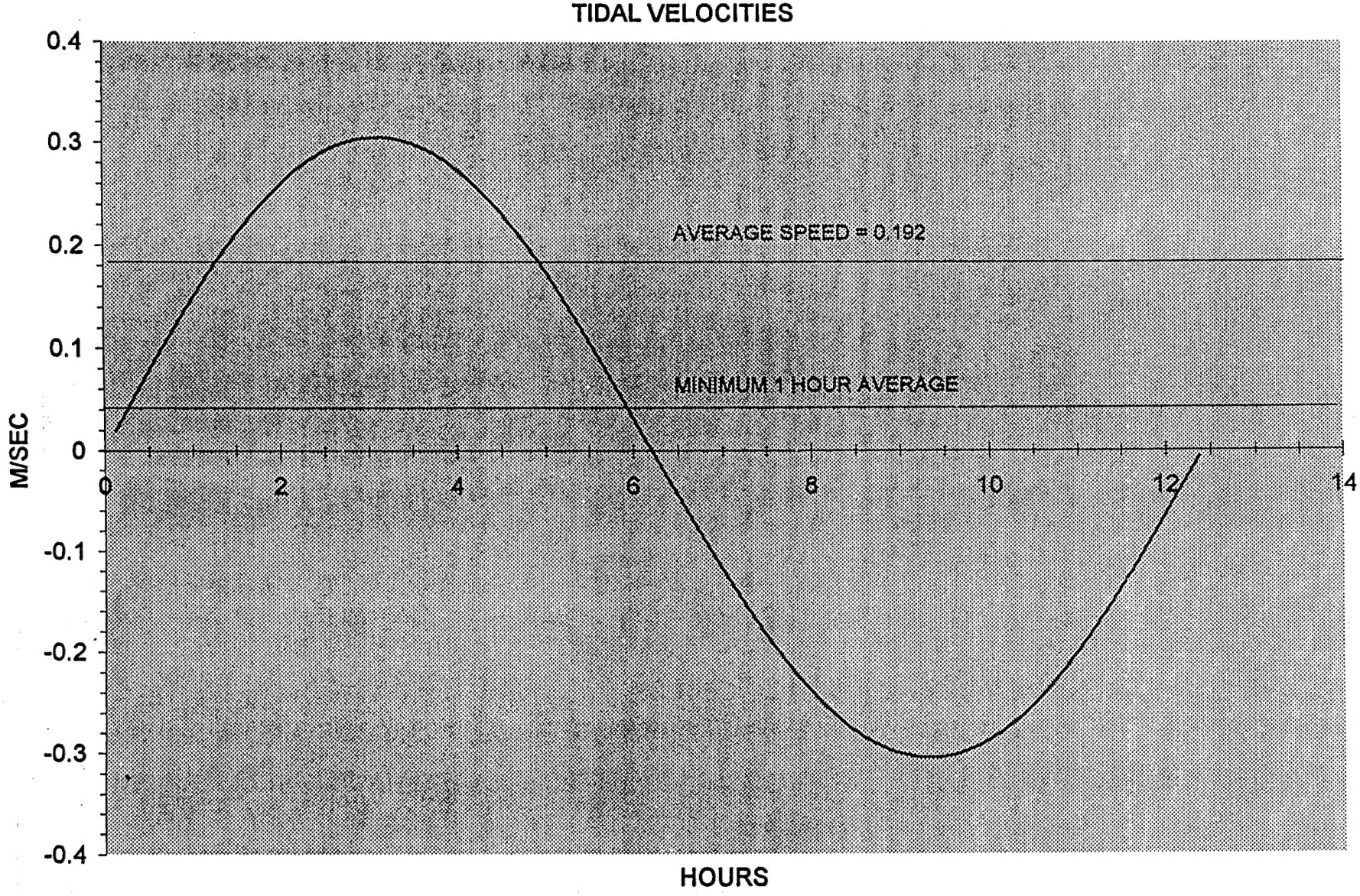
USEPA, 1990, "Technical Support Document for Water Quality-based Toxics Control," EPA-/505/2-90-001.

APPENDIX A

TIME	AMPL.	
(hours)	(m)	
	0	0
0.125	0.019274205	0.019274
0.25	0.038471361	0.038471
0.375	0.057514728	0.057515
0.5	0.07632818	0.076328
0.625	0.094836512	0.094837
0.75	0.112965735	0.112966
0.875	0.130643379	0.130643
1	0.147798777	0.147799
1.125	0.164363351	0.164363
1.25	0.180270883	0.180271
1.375	0.195457785	0.195458
1.5	0.209863346	0.209863
1.625	0.22342998	0.22343
1.75	0.236103455	0.236103
1.875	0.247833108	0.247833
2	0.25857205	0.258572
2.125	0.268277354	0.268277
2.25	0.276910221	0.27691
2.375	0.284436141	0.284436
2.5	0.290825031	0.290825
2.625	0.296051351	0.296051
2.75	0.300094208	0.300094
2.875	0.302937441	0.302937
3	0.304569684	0.30457
3.125	0.304984413	0.304984
3.25	0.304179969	0.30418
3.375	0.302159569	0.30216
3.5	0.298931289	0.298931
3.625	0.294508033	0.294508
3.75	0.288907484	0.288907
3.875	0.282152031	0.282152
4	0.274268677	0.274269
4.125	0.265288936	0.265289
4.25	0.255248705	0.255249
4.375	0.24418812	0.244188
4.5	0.232151395	0.232151
4.625	0.219186648	0.219187
4.75	0.205345703	0.205346
4.875	0.190683891	0.190684
5	0.175259821	0.17526
5.125	0.159135153	0.159135
5.25	0.142374342	0.142374
5.375	0.125044392	0.125044
5.5	0.107214578	0.107215
5.625	0.088956175	0.088956
5.75	0.07034217	0.070342
5.875	0.051446972	0.051447
6	0.032346116	0.032346



6.125	0.013115957	0.013116					
6.25	-0.006166633	0.006167	0.043252				
6.375	-0.025424572	0.025425					
6.5	-0.044580877	0.044581					
6.625	-0.06355897	0.063559					
6.75	-0.082282986	0.082283					
6.875	-0.100678077	0.100678					
7	-0.118670709	0.118671					
7.125	-0.136188955	0.136189					
7.25	-0.153162787	0.153163					
7.375	-0.169524352	0.169524					
7.5	-0.185208246	0.185208					
7.625	-0.200151771	0.200152					
7.75	-0.214295191	0.214295					
7.875	-0.227581968	0.227582					
8	-0.239958988	0.239959					
8.125	-0.251376774	0.251377					
8.25	-0.261789684	0.26179					
8.375	-0.271156092	0.271156					
8.5	-0.279438556	0.279439					
8.625	-0.286603967	0.286604					
8.75	-0.292623681	0.292624					
8.875	-0.297473635	0.297474					
9	-0.30113444	0.301134					
9.125	-0.303591463	0.303591					
9.25	-0.304834882	0.304835					
9.375	-0.304859727	0.30486					
9.5	-0.303665897	0.303666					
9.625	-0.301258166	0.301258					
9.75	-0.297646158	0.297646					
9.875	-0.292844312	0.292844					
10	-0.286871824	0.286872					
10.125	-0.279752568	0.279753					
10.25	-0.271515004	0.271515					
10.375	-0.262192061	0.262192					
10.5	-0.251821007	0.251821					
10.625	-0.240443302	0.240443					
10.75	-0.228104426	0.228104					
10.875	-0.214853706	0.214854					
11	-0.200744109	0.200744					
11.125	-0.18583204	0.185832					
11.25	-0.170177108	0.170177					
11.375	-0.153841896	0.153842					
11.5	-0.136891701	0.136892					
11.625	-0.119394284	0.119394					
11.75	-0.101419589	0.10142					
11.875	-0.083039469	0.083039					
12	-0.064327401	0.064327					
12.125	-0.045358184	0.045358					
12.25	-0.026207648	0.026208					
12.375	-0.006952348	0.006952					
	AVERAGE SPEED=	0.192927					


TIDAL VELOCITIES
0.4
0.3
0.2
0.1
0
-0.1
-0.2
-0.3
-0.4
M/SEC
AVERAGE SPEED = 0.192
MINIMUM 1 HOUR AVERAGE
0
2
4
6
8
10
12
14
HOURS

Estimated Discharge Rates in the Surficial Aquifer
Beach Point (Peninsula) Study Area
Aberdeen Proving Ground-Edgewood Area, Maryland

Assumptions: V_{hor} measured in MW-33B.1 is continuous throughout the Surficial Aquifer at the Beach Point peninsula; V_{hor} orientation (flow direction) is consistant with actual aquifer conditions over one complete tidal cycle (discounting short-term fluctuations due to tidal changes); the Surficial Aquifer at Beach Point is homogeneous, characterized by isotropic flow conditions.

MW-33B.1 (screened 41'-46'): V_{hor} = 0.89 ft/day (61.8 NE)
Average saturated thickness in Surficial Aquifer = 61.8'
Length of Beach Point peninsula fronting Bush River = 1,470'
Area of saturated Surficial Aquifer fronting Bush River = 90,882.7 ft^2

1. If we assume V_{hor} is oriented perpendicular (orthogonal) to the Bush River frontage in the Surficial Aquifer, then the estimated groundwater discharge from the Surficial Aquifer to the Bush River side of Beach Point can be calculated as shown below:

   Estimated Discharge Rate = (V_{hor})(Area of Surficial Aquifer X-Section fronting Bush River)
   = (0.89 ft/day)(61.8')(1,470')
   = 80,852.9 ft^3/day = 604,780.0 gallons/day

2. However, a more realistic estimation of discharge through the Surficial Aquifer along the Bush River involves using the directional component of V_{hor} orthogonal to the Bush River frontage. If this is performed, then the estimated groundwater discharge from the Surficial Aquifer to the Bush River side of Beach Point can be calculated as shown below:

   Estimated Discharge Rate = (V_{hor})(Area of Surficial Aquifer X-Section fronting Bush River)(component of V_{hor} orthogonal to Bush River frontage)
   = (0.89 ft/day)(61.8')(1,200.3')(sin 23.8) + (0.89 ft/day)(61.8')(270.3')
   = 41,508.7 ft^3/day = 310,484.8 gallons/day

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